1-1-2005

A comparison of seagrass communities at varying proximity to a low-density mussel-line aquaculture in King George Sound, Western Australia

John A. Eyres
Edith Cowan University

Edith Cowan University

Copyright Warning

USE OF THESIS

The Use of Thesis statement is not included in this version of the thesis.

A COMPARISON OF SEAGRASS COMMUNITIES AT VARYING PROXIMITY TO A LOW-DENSITY MUSSEL-LINE AQUACULTURE IN KING GEORGE SOUND, WESTERN AUSTRALIA.

John Alexander Eyres

(Bachelor of Science)

This thesis is presented for the award of Master of Science

Edith Cowan University

Faculty of Communication, Health and Science

November 2005

COPYRIGHT AND ACCESS

ABSTRACT

Increasingly, mussel-line aquaculture is recognised as a potential threat to seagrass. Sites suitable for mussel-line aquaculture are often in sheltered waters containing seagrass. Despite this, few studies have examined the risk of mussel-line aquaculture to seagrass ecosystems. The objective of this study is to determine how low-density, mussel-line aquaculture might influence the underlying seagrass ecosystem.

Sampling took place in summer and spring, 2000, at sites located along two transects running 500 m from the centre of a low-density mussel farm (1.3 t of mussels/Ha), over *Posidonia sinuosa* meadows at Misery Beach, Albany, Western Australia. This site is well-flushed and dispersal by strong currents would limit any impact of mussel waste. A range of seagrass epibenthic and infauna, epiphytic macroalgae, water-column and nutrients parameters were measured at the centre of the farm, and then west and east along each transect, at the edge of the farm, 100 m and 500 m from the edge of the farm. The strongest patterns in the measured variables were observed in summer. At this time, ordination and ANOSIM showed a gradient in assemblage structures of epibenthic macrofauna coinciding with distance from the farm. Assemblages 500 m from the farm were different from those at the centre. Assemblages at sites on the edges and 100 m from the farm were not different from each other, and were intermediate in structure compared to the centre and 500 m sites. Densities of epifauna were greatest at sites 100 m from the farm, were lowest at sites at the centre and edges of the farm, and were also low at sites 500 m away from the farm. Benthic infauna were absent at sites at the centre of the farm and sites adjacent the farm on the west. Infauna assemblages were most similar among sites close to the farm and were most dissimilar among sites that were more distant from each other. In the summer, densities were greatest at sites 100 m from the farm, extremely low beneath the farm, and intermediate between these at sites 500 m away from the farm. Seagrass shoot densities were highest at sites 500 m from the farm. Trends for other seagrass variables, although not statistically significant, suggest an increase near the farm. Epiphyte biomasses (AFDW) were higher in the centre, on the edges and 100 m from the farm, than at the sites 500 m from the farm. Ammonium concentrations in the porewater were higher at sites on the edge of the farm than at the sites 500 m from the farm, though there was no clear difference in loss of ignition of the sediment. In summer and spring, chlorophyll-a concentrations in the sediment were not significantly different between sites, although weak and insignificant trends reflected observed patterns for infauna densities. Most variables displayed similar but less pronounced trends in spring than in summer. However, high densities of epifauna beneath the farm in

spring, contrasted to observed low densities observed beneath the farm during summer, despite similar peak densities of epifauna observed adjacent to the farm.

The findings of this study are consistent with a benthic pathway of effect of aquaculture on seagrass ecosystems. It is suggested that mussel-line aquaculture has resulted in elevated porewater ammonium concentrations near the farm and efflux of this ammonium from the sediments stimulated epiphyte growth, with subsequent effects on shoot densities, at sites near the farm. In turn, functions of seagrass and epiphytes as habitat and food may have been altered, influencing epibenthic macrofauna and infauna assemblages. Epifauna and infauna compositions were affected by proximity to the farm. Less consistent effects were noted on densities, although generally densities were lower at sites 500 m from the farm than sites near the farm, corresponding to the trend for epiphyte biomasses and inverse of the trend for shoot densities. The study shows that at these stocking densities, which were lower than a typical commercial farm, the effects of mussel-line aquaculture appear to be subtle. The effects of higher stocking densities on mussel-lines, or the effects of farms in less well-flushed environments can not be inferred from the results of this study, however, it is reasonable to assume that any effects would be more significant than those observed in this study of a low-density farm in a well-flush environment.

DECLARATION

I certify that this thesis does not, to the best of my knowledge and belief, incorporate without acknowledgment, any material previously submitted for a degree or diploma in any institution of higher education; contain any material previously published or written by another person except where due reference is made in the text; or contain any defamatory material.

Signed. 25/11/05

ACKNOWLEDGEMENTS

I would like to thank my Supervisors, Assoc. Prof. Paul Lavery and Dr Glenn Hyndes of Edith Cowan University, for their guidance, wisdom and patience over the years.

I would also like to thank all who assisted me with my field work:
Tim Carr, David Eyres, Pam Haddon, Daniella Hanf, Michael Harrison, Justin King, Marcus Kort, Mark Luther, Peter Quintana, Andrew Richardson, Alex Watson.

A special thanks goes to David Eyres, Daniella Hanf, Marcus Kort, Peter Quintana, Alex Watson, without whom the field-work would not have been possible.

Finally, I would like to thank my mum, Christine Eyres for her support, and for the production of six high quality epibenthic macrofauna nets, and Alex Watson for his suggestions and for being a good mate, and my dad, David Eyres, for all of his help over the years.

I would like to dedicate this thesis to Kelly Scaramozzino for her encouragement during this project, her contribution of many high quality diagrams of epibenthic macrofauna taxa, and for her tolerance to coming second to this project, many times.

CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1: INTRODUCTION

1.1 OPENING

Many people believe that aquaculture production will relieve pressure on wild fisheries, however, there is little evidence of this (Naylor et al. 2000). It has even been stated that aquaculture could overtake cattle grazing as a food source by the end of this decade (Brown 2000). Such projections are based on world aquaculture production increasing at a rate of 9.2% per year since 1970, which is more rapid than any other animal food producing sector (F.A.O. 2002). The contribution aquaculture makes to global supplies of fish, crustaceans and molluscs increased from 3.9% of the total production mass in 1970 to 27.3% by 2000.

Of the reported total aquaculture production in 2000 (45.7 million tonnes), 23.5% were molluscs (F.A.O. 2002). Relative to other forms of aquaculture, shellfish aquaculture is often considered benign or even beneficial. Several studies have shown that extensive aquaculture of shellfish in polyculture systems can be used to reduce particulate loads, thereby mitigating the impact of more intensive aquaculture in the same area (Naylor et al. 2000). However, this does not demonstrate that shellfish aquaculture is environmentally benign. Indeed, several studies have recorded substantial impacts to local, non-vegetated environments (Tenore et al. 1982, Mattsson & Linden 1983, Hatcher et al. 1994, Stenton-Dozey et al. 2001). Such impacts could extend to vegetated environments. Seagrass meadows are the foundation of many natural resources and are important to global marine diversity (Short & Wyllie-Echeverria 1996, Kirkman 1997), but, to date, very few studies have examined the effects of shellfish aquaculture on seagrass.

Mussel farming is an expanding industry in Western Australia. However, sites suitable for mussel aquaculture are commonly located in sheltered waters, which often contain seagrass meadows. While only one farm is currently located over seagrass, any expansion of this industry is likely to result in more farms being placed above seagrass ecosystems, which could result in an impact on those systems. Production by 14 growers, rose from 659 tonnes in 1997/1998 to 964 tonnes by 2000/2001, and is expected to continue growing (Maguire 2002).

1.2 AIM

This research aims to identify and quantify the impacts of mussel-line aquaculture on a seagrass ecosystem. More specifically, the aim is to describe any effects of mussel-line aquaculture on seagrass, epiphytes, epibenthic macrofauna and macroinfauna within a *Posidonia sinuosa* community. The aim will be achieved by answering the question: Do spatial trends in selected measurable variables representing ecological functions in *Posidonia sinuosa* meadows correspond to the location of a mussel-line aquaculture lease? This study describes the effects of a mussel-line aquaculture with a stocking density of approximately 20 tonnes suspended over 15 hectares of a seagrass ecosystem in a well-flushed environment. It will contribute to an understanding of the processes within seagrass ecosystems, with particular reference to those affected by mussel-line aquaculture. The study also provides local mussel producers and regulatory authorities with qualitative data on the possible impacts of mussel-line aquaculture on seagrass ecosystems.

1.3 SIGNIFICANCE

Western Australian mussel growers in Albany and Cockburn Sound are large contributors of *Mytilis edulis planulatus* to the Australian market (Maguire 2002). The need to undertake research arose when local mussel farmers in Albany, Western Australia, sought approval to expand their mussel-line operations. The proposed expansion would encroach on seagrass ecosystems, as the only suitable sites for mussel aquaculture are located in the sheltered waters of King George Sound, which contain extensive meadows of the seagrass *Posidonia sinuosa* (1999). In response to the proposal, the Department of Environmental Protection requested that a comprehensive, independent and scientific study be undertaken into the impacts of mussel-line aquaculture on local seagrasses (Albany Harbours Planning Committee, 1999). The study neglected the perceived risk to seagrass, as the impact of mussel aquaculture on any species of seagrass anywhere in the world had not been documented. All previous studies of its impacts were in non-vegetated benthic ecosystems (Mattsson & Linden 1983, Kaspar et al. 1985, 1987, Baudinet 1990, Hatcher et al. 1994, Grant et al. 1995, 1999, La Rosa et al. 2000, Mirto et al. 2000, Chamberlain et al. 2001, Stenton-Dozey et al. 2001). While not undertaken as part of any approval process, this study was initiated to contribute to the limited knowledge base on the effects of mussel aquaculture on seagrass communities. The research is significant, as few studies have investigated the environmental impact of aquaculture in Australia, and the study contributes to the very limited knowledge concerning how Australian seagrass ecosystems interact with mussel-line aquaculture culture methods.

1.4 BACKGROUND

Aquaculture in the marine environment covers a wide range of farming practices, each type uniquely interacting with the environment (Iwama 91). Such interactions can result in impacts on the environment, usually through physical disturbance and the output of by-products. The extent of those impacts depends on the biological and physicochemical characteristics of the particular ecosystem (Black 2001). Despite its various forms, the effects of aquaculture are largely influenced by the practice of feeding (Beveridge et al. 1994). Aquaculture can be divided into four major types based on the method of cultivation; extensive, semi-intensive, intensive, and super-intensive.

Intensive aquaculture, such as finfish and shrimp aquaculture, refers to the high density cultivation of organisms, where the environmental conditions are manipulated throughout the life-cycle. Due to high stocking densities, the organisms rely on the addition of food to prevent starvation, and antibiotics to prevent disease (Beveridge et al. 1994). The diets are artificially formulated for maximum growth rates and minimal wastage, however a proportion of feed inevitably settles on the benthos (Black 2001). Most of the more commonly cited examples of aquaculture that have caused environmental impacts, are forms of intensive aquaculture (Iwama 1991). Recently, the term "*superintensive*" has been applied to the land-based aquaculture of organisms at maximum stocking densities (Midlen & Redding 1998). Superintensive aquaculture is only possible using rapid recirculating systems with highly efficient water filters, within an environment that is completely controlled using a high level of technology, including the comprehensive formulated diet required for the particular species (Midlen & Redding 1998, Appleford et al. 2003).

The density of organisms is lower for *semi-intensive* aquaculture, thus organisms require less manipulation of the environmental conditions, and any feeding serves only to supplement the natural diet of the organisms (Pillay 1992, Swift 1993, Appleford et al. 2003). In comparison, *extensive* aquaculture cultivates organisms in an environment similar to the natural habitat. This type of aquaculture therefore has a low ratio of stock to unit area, and does not supplement nutrition (Appleford et al. 2003). Mussel aquaculture is a form of extensive aquaculture, with minimum husbandry work needed as mussels acquire nutrition by filtering phytoplankton (Swift 1993). Mussels are one of the most efficient groups of animals at converting food into biomass (Swift 1993). Mussel aquaculture infrastructure is relatively simple, requiring little investment. The colonies of mussels are commonly suspended at the top of the water column, where phytoplankton is usually in greatest concentrations (Swift, 1993). Mussel aquaculture is restricted to coastal areas where the infrastructure is sheltered from damaging weather

conditions. However, water flow must be sufficient to provide enough food and oxygen to the mussels, and water quality should be high to ensure the mussels are fit for human consumption (Beasley & Maguire 2000).

1.5 IMPACTS OF AQUACULTURE

It is well recognised that the impacts of aquaculture are highly dependent on the type of cultivation, and the local environmental factors (Iwama 1991, Pillay 1992, Beveridge et al. 1994, Grant et al. 1995, Hargrave et al. 1997, Mendez et al. 1997). Occasionally, the environmental impacts of aquaculture are minimal and sometimes positive (Beveridge et al. 1994). However, the effects of aquaculture are usually negative and associated with consumption of resources and production of wastes, which can lead to changes in the ecosystem (Black 2001). Small areas of water are used to cultivate unnaturally large populations, resulting in elevated rates of organic loading, habitat destruction, shading and deoxygenation of the water column (Pillay 1992, Beveridge et al. 1994, Grant et al. 1995, De Casabianca et al. 1997a, Hargrave et al. 1997, Delgado et al. 1999). Of these, organic enrichment is potentially the most significant problem and is associated with bacterial decomposition that can result in oxygen depletion, nutrient enrichment of the benthos, and smothering of benthic organisms by sediment. Faunal community structure is known to alter with varying degrees of organic enrichment (Weston 1990). The effects of organic enrichment is commonly associated with benthic community changes, such as reduced production, changes in trophic dominance and loss of biodiversity (Pearson & Rosenburg 1978, Weston 1990, De Casabianca et al. 1997a, Findlay 1997, Hargrave et al. 1997).

The hydrodynamics of an environment strongly modulate the effects of aquaculture, both in terms of organic loading and its influence on the capacity of the environment to balance the input and output of nutrients and gasses (Iwama 1991). The dilution and distribution of waste products directly relate to the environment's holding capacity for aquaculture in relation to the type and density of stock (Pearson & Rosenburg 1978, Wu 1994, Findlay & Watling 1995, Findlay 1997, Hargrave et al. 1997, Henderson et al. 2001).

The impact of organic enrichment associated with semi-intensive aquaculture, such as finfish aquaculture, on benthic ecosystems is well documented (Silvert 1992, Beveridge et al. 1994, Henderson 1995, Wu 1995, Hargrave et al. 1997, Karakassis & Hatziyanni 2000). Structural changes in macrofauna communities have been observed, where inputs of feed and faecal matter have resulted from fish farming. Particular taxa of benthic organisms are important indicators of

ecosystem degradation as they display consistent patterns of change within a broader community structure at various degrees of environmental disturbance (Pearson & Rosenburg 1978, Clarke & Warwick 1994). For example, polychaetes have been shown to respond to increasing organic enrichment, with reduced species richness and increased abundance of particular taxa (e.g. *Capitella capitata*) in more enriched environments. (Pearson & Rosenburg 1978, Weston 1990, Henderson 1995, Findlay 1997, Hargrave et al. 1997, Cardell et al. 1999, Samuelson 2001). The impact of organic enrichment from aquaculture on benthic communities is due partly to the influence of storm related re-suspension of the sediments (Findlay & Watling 1995), but can be highly seasonal (Yokoyama 2002).

Most studies that have examined the impact of aquaculture on benthic communities took place in unvegetated habitats. Although several studies have examined the impacts of finfish aquaculture in vegetated ecosystems, the emphasis has been on the loss and degradation of benthic macrophytes (Mendez et al. 1997, Delgado et al. 1999, Dimech et al. 2000, Ruiz et al. 2001). Literature regarding the effects of aquaculture on the broader communities within vegetated ecosystems remains limited. Furthermore, the environmental effects of extensive aquaculture have not been extensively documented, and although it is the type of cultivation that facilitates the least organic deposition, it too can be associated with environmental change (Iwama 1991).

1.6 IMPACTS OF MUSSEL AQUACULTURE

As opposed to intensive finfish aquaculture, mussel cultivation has relatively low impact, because it does not result in a net addition of nutrients to the environment. For example, Folke and Kautsky (1989) found that sedimentation increased twenty-fold under a 40-ton finfish aquaculture, while under a 100-ton mussel aquaculture it increased only threefold. It was estimated that intensive finfish aquaculture was approximately 15 times more harmful to the benthos than mussel aquaculture (Folke & Kautsky 1989). Despite this, potential impacts of mussel farms include organic enrichment and physical disturbance of the benthos and hydrography. Although mussels do not require artificial feeding, they do consume substantial quantities of suspended matter from the water column, mainly phytoplankton. Mussels are agents of sedimentation and nutrient cycling because a large percentage of ingested material is egested in the form of faeces and pseudofaeces (Kautsky & Evans 1987) that settles at the benthos, is available to benthic consumers and decomposed by bacteria (Fabiano et al. 1994). Mussels also release a fraction of their waste as dissolved nutrients, to the water column as dissolved nutrients (Kautsky & Evans 1987). Therefore, mussels play a major role in the

conversion of ambient, suspended, particulate organics, into bioavailable nutrients, such as nitrate, nitrite and ammonium, which accumulate locally.

Although mussel cultivation can potentially improve general water quality via the consumption of phytoplankton (Black 2003), and thus lead to nutrient loss from the system during harvesting (Kaspar et al. 1985), several studies have shown that mussel colonies can be a source of nutrients to the immediate environments (Kaspar et al. 1985, Asmus & Asmus 1991). Mussels are known to accelerate processes by which the mineralisation of organic material allows nutrients to become bioavailable (Beveridge et al. 1994, Gilbert 1997). Due to mussels ingesting organic particles other than phytoplankton, a mussel colony has the potential to stimulate a net increase in phytoplankton biomass (Asmus & Asmus 1991). Ingested particulate organic nitrogen can be returned to the water as dissolved inorganic nitrogen thereby becoming bioavailable to phytoplankton (Josefsen & Schluter 1994). Many subsequent studies have established that mussel colonies regenerate nitrogen and are net producers of dissolved inorganic nitrogen (Gibbs et al. 1992, Prins & Smaal 1994, Smaal & Zurburg 1996, Barranguet 1997, Gilbert 1997, Ogilvie et al. 2000).

Increased levels of bioavailable nutrients in the water column can stimulate phytoplankton blooms and epiphytic algae that devastate benthic communities by blocking light (Cambridge & McComb 1984, Cambridge et al. 1986, Silberstein et al. 1986). It is therefore possible that mussel aquaculture could affect the benthos through a similar pelagic mechanism of cause-effect. However, it is not clear which pathways of cause-effect from mussel aquaculture are most threatening to ecosystems exposed to mussel aquaculture. While a pelagic pathway of cause-effect is possible, there are few examples available.

Ammonium was the most significant biochemical response from the sediment beneath a mussel farm in Nova Scotia, Canada (Hatcher et al. 1994). While sediment at sites away from the farm were a net sink for total dissolved nitrogen, the sediment beneath the farm was shown to be a source of ammonium all year around (Hatcher et al. 1994). Microbial activity in organically enriched sediments can result in anoxia (Mattsson & Linden 1983, Mirto et al. 2000, Stenton-Dozey et al. 2001) and has facilitated the mineralisation of nutrients, most importantly, ammonium from the sediment (Hatcher et al. 1994, Grant et al. 1995).

Apart from pseudofaeces and faeces contributing to organic enrichment of sediments beneath mussel farms, whole dead mussels and gametes are also a potential source of nutrient loading to the benthos (Fabiano et al. 1994, Grant et al. 1995). Nutrient release from decomposing somatic tissue of mussels has been recorded as being approximately 10 times higher than the rate of

conversion of ambient, suspended, particulate organics, into bioavailable nutrients, such as nitrate, nitrite and ammonium, which accumulate locally.

Although mussel cultivation can potentially improve general water quality via the consumption of phytoplankton (Black 2003), and thus lead to nutrient loss from the system during harvesting (Kaspar et al. 1985), several studies have shown that mussel colonies can be a source of nutrients to the immediate environments (Kaspar et al. 1985, Asmus & Asmus 1991). Mussels are known to accelerate processes by which the mineralisation of organic material allows nutrients to become bioavailable (Beveridge et al. 1994, Gilbert 1997). Due to mussels ingesting organic particles other than phytoplankton, a mussel colony has the potential to stimulate a net increase in phytoplankton biomass (Asmus & Asmus 1991). Ingested particulate organic nitrogen can be returned to the water as dissolved inorganic nitrogen thereby becoming bioavailable to phytoplankton (Josefsen & Schluter 1994). Many subsequent studies have established that mussel colonies regenerate nitrogen and are net producers of dissolved inorganic nitrogen (Gibbs et al. 1992, Prins & Smaal 1994, Smaal & Zurburg 1996, Barranguet 1997, Gilbert 1997, Ogilvie et al. 2000).

Increased levels of bioavailable nutrients in the water column can stimulate phytoplankton blooms and epiphytic algae that devastate benthic communities by blocking light (Cambridge & McComb 1984, Cambridge et al. 1986, Silberstein et al. 1986). It is therefore possible that mussel aquaculture could affect the benthos through a similar pelagic mechanism of cause-effect. However, it is not clear which pathways of cause-effect from mussel aquaculture are most threatening to ecosystems exposed to mussel aquaculture. While a pelagic pathway of cause-effect is possible, there are few examples available.

Ammonium was the most significant biochemical response from the sediment beneath a mussel farm in Nova Scotia, Canada (Hatcher et al. 1994). While sediment at sites away from the farm were a net sink for total dissolved nitrogen, the sediment beneath the farm was shown to be a source of ammonium all year around (Hatcher et al. 1994). Microbial activity in organically enriched sediments can result in anoxia (Mattsson & Linden 1983, Mirto et al. 2000, Stenton-Dozey et al. 2001) and has facilitated the mineralisation of nutrients, most importantly, ammonium from the sediment (Hatcher et al. 1994, Grant et al. 1995).

Apart from pseudofaeces and faeces contributing to organic enrichment of sediments beneath mussel farms, whole dead mussels and gametes are also a potential source of nutrient loading to the benthos (Fabiano et al. 1994, Grant et al. 1995). Nutrient release from decomposing somatic tissue of mussels has been recorded as being approximately 10 times higher than the rate of

nutrient release from living mussels (Fabiano et al. 1994). Fallen mussels, rather than biodeposition, were recorded as having the greatest impact on benthic macrofauna (Grant et al. 1995).

High rates of biodeposition and organic enrichment of the benthos have altered the biochemistry of the sediments near mussel farms, usually resulting in changes to benthic communities and in anaerobic conditions beneath and around mussel farms (Tenore et al. 1982, Mattsson & Linden 1983, Mirto et al. 2000, Stenton-Dozey et al. 2001). Benthic faunal assemblages have been shown to reflect trends in organic enrichment over distances away from mussel aquaculture (Kroncke 1996). Areas close to the farm are commonly associated with decreased taxa diversity of macrofauna, and often, increased abundance of opportunistic polychaetes such as *Capitella capitata* (Tenore et al. 1982, Mattsson & Linden 1983, Mirto et al. 2000, Stenton-Dozey et al. 2001).

1.7 SEAGRASS ECOSYSTEMS: IMPORTANCE AND THREATS

Seagrass meadows are important sinks for organic and inorganic nutrients and form the basis for detrital food webs in which nutrients and gasses are recycled (Valiela 1995). They provide structurally complex habitats for the juveniles of many species of fauna (Short & Wyllie-Echeverria 1996, Hannan & Williams 1998), and are a resource base within ecosystems that rank among the most productive in the world (Kirkman, 1997). Seagrasses form physical barriers that slow water currents, help to filter suspended sediments from the water column and stabilise the bottom sediments (Kirkman et al. 1991, Kirkman 1997). For all of the above reasons, seagrasses are important contributors to global marine biodiversity and can form important habitats for economically valuable fisheries (Hill 1993, Jernakoff et al. 1993).

Degradation of coastal benthic ecosystems has been extensive within Australia (Walker & McComb 1992), with significant loss of seagrass attributable to anthropogenic pollution of coastal environments (Kirkman 1997). Seagrass degradation has occurred through direct destruction, oxygen depletion and nutrient enrichment (Gordon et al. 1994, Short & Wyllie-Echeverria 1996). It is well known that seagrass meadows are susceptible to reduced light intensity, resulting from phytoplankton blooms and epiphytic loading which accompanies nutrient enrichment (Cambridge & McComb 1984, Cambridge et al. 1986, Silberstein et al. 1986). This often leads to seagrass death and community change. Aquaculture has been recognised as one of many potential threats to seagrasses. Several studies have assessed the impact of finfish aquaculture on seagrass communities around the Mediterranean, most

reporting significant reductions or complete mortality of seagrass beneath and near the cages (Mendez et al. 1997, Delgado et al. 1999, Pergent et al. 1999, Dimech et al. 2000, Ruiz et al. 2001). In addition, high epiphyte biomass has been reported below and near fish farms (Delgado et al. 1999, Dimech et al. 2000). Delgado et al. (1999) suggest that organic material that built up in the sediment close to the farm was linked to changes in the surrounding seagrass, while Dimech (2000) notes that suspended organic matter and phytoplankton may have been responsible for the reduced photosynthetic capacity of seagrass near a fish farm. Ruiz (2001) attributed significant loss and degradation of *Posidonia oceanica* to an increase in grazing pressure, due to the presence of a farm. The significance of the herbivory in this case was thought to be dependent on complex interactions between many processes, including reduced water transparency and increased dissolved nutrients and organic content of the sediment.

1.8 POTENTIAL IMPACTS OF MUSSEL AQUACULTURE ON SEAGRASS ECOSYSTEMS

Vegetated ecosystems can be vastly different from non-vegetated ecosystems in their processes and functions (Valiela 1995, Connolly 1997), therefore mussel aquaculture could affect seagrass ecosystems differently. Seagrasses occupy space, shade the benthos, utilise nutrients and carbon dioxide, exude oxygen to the sediment surrounding the roots, provide structurally complex habitats, and can affect the flow of water and particles through the environment (Kaspar et al. 1985, Kautsky & Evans 1987, Hatcher et al. 1994). Seagrass meadows can influence food webs on a number of levels and have additional biochemical processes operating across the sediment-water interface (Hemminga et al. 1991). These processes potentially alter the affects of an aquaculture on an ecosystem. Anoxic conditions are typical of many temperate seagrass sediments (Filskov 1990, Hemminga 1998), due to the natural accumulation and microbial decomposition of organic material in the sediment. However, oxygen exudation from seagrass roots can enrich pockets of sediment surrounding the roots (Short 1986, Filskov 1990, Holmer 1992, Martinova 1993, Hemminga 1998, Connell & Walker 2001), allowing nitrifying bacteria to convert organic compounds containing nitrogen into nutrients such as ammonium, nitrates and nitrites, which are available for primary production. Therefore, oxygen exudation from seagrass roots could lower the biochemical oxygen demand in pockets of sediment, reducing the potential for hypoxia associated with mussel farm biodeposition. At the same time, nutrient absorption by seagrass may reduce the effects of nutrient enrichment on other organisms. For the reasons above, some of the chemical processes caused by a low intensity aquaculture may be counteracted by the processes within seagrass ecosystems, suggesting that seagrass meadows

potentially have a degree of resilience before a threshold is reached where aquaculture will cause an impact.

In a study of seagrass beneath and away from an oyster farm, Crawford (2003) found no clear signs of organic enrichment, or differences in benthic macrofauna. However, being an isolated study, these results cannot be extrapolated to aquaculture of other organisms, such as mussels, or different ecosystems. It is recognised that mussels facilitate the conversion of inaccessible particulate organic nutrients in the water column to nutrients available for absorption by macrophytes (Peterson & Heck Jr 1999). Thus, potentially, mussel aquaculture could affect seagrass communities, but interestingly, seagrass meadows could influence the processes that usually occur beneath mussel farms. Seagrass meadows could facilitate the retention of deposited faecal material, and leaked oxygen from the roots may facilitate oxidisation of faecal ammonium to highly soluble nitrate (Asmus et al. 1994, Mazouni 1996). Biodeposition can be converted by nitrifying bacteria to bioavailable nutrients, which can stimulate benthic flora. Therefore, epiphytes loads could smoother seagrass causing structural changes in seagrass, which can lead to meadow loss. As seagrasses morphology alters, epiphyte assemblages could change, potentially reducing the ability of seagrass to provide food and habitat to associated macrofauna, influencing benthic assemblages (Peterson & Heck Jr 2001b, Bostrom et al. 2002).

However, *Posidonia sinuosa* meadows can be nutrient limited, thus elevated nutrient concentrations within the rhizosphere could promote seagrass productivity (Martinova 1993, Reusch et al. 1994, Peterson & Heck Jr 1999, Touchette 2000, Peterson & Heck Jr 2001a, b, Stenton-Dozey et al. 2001, Garcia et al. 2002), which may affect the structure of seagrass differently to the response of seagrass to epiphyte loads. For example, the seagrass leaves may extend at a faster rate, decreasing the time for epiphyte assemblages to diversify.

Faeces, pseudofaeces, gametes, dead mussel and fowling organisms, derived from mussel aquaculture (Kautsky & Evans 1987) could directly or indirectly alter habitats of benthic macrofauna, and is available to particular guilds of benthic macrofauna, as a food source. Mussel biodeposition may advantage particular types of benthic macrofauna over others. Therefore, it could influence macrofauna assemblage structures associated with seagrass meadows. In addition to organic input to the benthos, the establishment of a farm can cause physical disturbance to the benthos. This can cause blow-outs and lead to erosion of seagrass meadows. Mussel-lines slow water currents, a process known to result in higher settlement rates of suspended material, such as phytoplankton, to the benthos (Pillay 1992). Settling organic material is decomposed by microbes, leading to an increase in dissolved nutrients, which accumulate locally and promote the growth of epiphyte and phytoplankton. Although, shading

from mussel-lines is minimal in comparison with other types of aquaculture structures (Black 2001), they potentially contribute to reductions in light at the benthos (Barranguet et al. 1996). Any decrease in light could exacerbate stress to seagrass already light inhibited.

Connell's classic paper of 1978, "Intermediate Disturbance Hypothesis" has been widely applied and developed to explain patterns in species diversity of many ecosystems (Dial & Roughgarden 1998). According to the intermediate disturbance hypothesis, the diversity of a benthic community will be greatest when disturbance is intermediate (Connell 1978). If the intensity or frequency of the disturbance is high, only pioneering species that colonise soon after the disturbance will be present and the diversity of the community will be low (Pearson & Rosenburg 1978). Similarly, if the intensity or frequency of the disturbance is very low, the community will reach climax, and species adapted to the climax state will competitively exclude other species, thereby reducing diversity (Pearson & Rosenburg 1978).

Compared to communities that are highly disturbed due to it being situated directly under a mussel aquaculture, communities situated on the edge of the mussel farm are potentially exposed to a more intermediate disturbance from biodeposition. According to Connell's hypothesis, an intermediate level of stress could disadvantage particular taxa, but would not totally remove them from the location. Therefore, some individuals would survive, and recolonisation would start, not only, from propagules or larvae, but also survivors. However the Intermediate Disturbance Hypothesis assumes moderate to high settlement of propagules or larvae, because of its important for recolonisation (Dial & Roughgarden 1998). Since water currents flow through them, aquaculture structures are known to increase settlement of particles, such as propagules and larvae. Therefore, the Intermediate Disturbance Hypothesis is relevant and could be plausible within the context of mussel farm impacts.

High densities of opportunistic species are characteristic of areas exposed to moderate organic enrichment (Pearson & Rosenburg 1978), and mussel aquaculture commonly causes these conditions in surrounding environments. If conditions due to moderate organic enrichment are sufficiently stressful to the dominant species of the area, and recruitment of a subdominant species can occur, then it is plausible that new species could take advantage of resources previously utilised only by the dominant species. As a result, intermediate levels of disturbance could encourage coexistence, which essentially increases species diversity (Dial & Roughgarden 1998). For this reason, sites near mussel aquaculture with high species diversity are potentially influenced by moderate organic enrichment.

1.9 RESEARCH OBJECTIVES

The broad objective of this study was to determine whether any differences in a seagrass ecosystem corresponded to the proximity of a mussel-line aquaculture. To achieve this, the literature was reviewed, enabling me to select various ecological processes for building two conceptual pathway models of cause-effect from mussel aquaculture on seagrass communities. Trends in selected variables were examined to establish whether benthic and/or pelagic pathways explained any cause-effect of a mussel aquaculture on seagrass communities. The two pathways are not mutually exclusive, and the decision as to which variables were measured for the study was based on whether they could explain a benthic and/or a pelagic pathway. The benthic pathway model, explained in section 1.10, starts with mussels in aquaculture enriching the benthos with organic nutrients, which can be bacterially remineralised to bioavailable nutrients, which stimulate benthic flora, and consequently influences associated macrofauna. The pelagic pathway model, explained in section 1.10, starts with mussels in aquaculture releasing waste and dissolved nutrients into the water column. The waste decreases water clarity and the dissolved nutrients stimulate phytoplankton growth; both variables reducing light at the benthos, leading to changes in seagrass communities. From the review, specific variables that represent ecological processes, observed within seagrass communities and the overlying water column, at varying distances from the mussel-line aquaculture. The ultimate aim of the study was to determine whether trends in those variables correspond to proximity to the mussel aquaculture, and fit within the context of one of the proposed pathway models that link mussel aquaculture to changes in seagrass communities.

1.10 CONCEPTUAL PATHWAYS OF CAUSE-EFFECT

An outline of the two main cause-effect pathways of mussel-line aquaculture on seagrass systems are provided in Figure 1.1, 1.2 and 1.3. Mussel colonies in aquaculture ingest large quantities of phytoplankton and other organic particles, which is egested in the form of faeces and pseudofaeces (Kautsky & Evans 1987). In the pelagic model, the faecal material, while suspended in the water column, could block some of the light to the benthos, directly decreasing water clarity (Figure 1.1). Dissolved nutrients excreted by mussels, such as nitrate, nitrite and ammonium could indirectly decrease water clarity, by promoting phytoplankton production in the water column, thereby blocking light to the benthos (Asmus & Asmus 1991). A reduction in light to the benthos, through increased light attenuation, could reduce the capacity of seagrass to photosynthesise. This could alter seagrass attributes, such as seagrass density and biomass,

which are known to influence associated fauna, via habitat changes (Bell & Westoby 1986, Edgar 1990, Connolly & Butler 1996, Jernakoff & Nielsen 1997a, Webster 1998, Sanchez-Jerez et al. 2000, Lee et al. 2001, Hovel et al. 2002).

In the benthic model (Figure 1.2), mussels ingest phytoplankton and other organic practicals, which is egested in the form of faeces and pseudofaeces (Kautsky & Evans 1987). Dead mussels and gametes, though reproduction and mortality of mussels, and fowling organisms, through farm maintenance, also contributes to the waste, which settles at the benthos. The three-dimensionality of mussel shells can provide refuge or habitat to particular macrofauna taxa, and faecal material could settle at the benthos reducing the average particle size of the sediments. Therefore organic waste could directly alter habitats of benthic macrofauna, and provide an available food source to particular guilds of benthic macrofauna, directly influencing faunal assemblage structures. However, mussel waste could also indirectly affect benthic macrofaunal assemblage structures. As mussel waste is incorporated into the sediment. The organic waste is potentially an available food source to an array of benthic consumers, and could directly alter their habitats, thus assemblage structures of benthic fauna could change. As it is incorporated into the sediment the mussel waste is colonised by microbes including denitrifying bacteria (Fabiano et al. 1994), which leads to remineralisation of nutrients, and consumption of oxygen via respiration (Mazouni 1996). Any depletion of oxygen in the sediment or at the sediment-water interface, due to microbial respiration, could provide considerable stress to many benthic macrofauna taxa, altering assemblage structures. Remineralised nutrients within the sediment porewater could facilitate changes in seagrass and microphytobenthos, thus affecting benthic macrofauna through altered habitat or through food provision. Some of the remineralised nutrients in the sediments could flux to the water column through a benthic-pelagic pathway, promoting epiphyte growth on seagrass leaves, thereby reducing the capacity of seagrass to photosynthesise (Figure 1.3). This could affect seagrass associated macrofauna, via habitat alterations and changes in food availability.

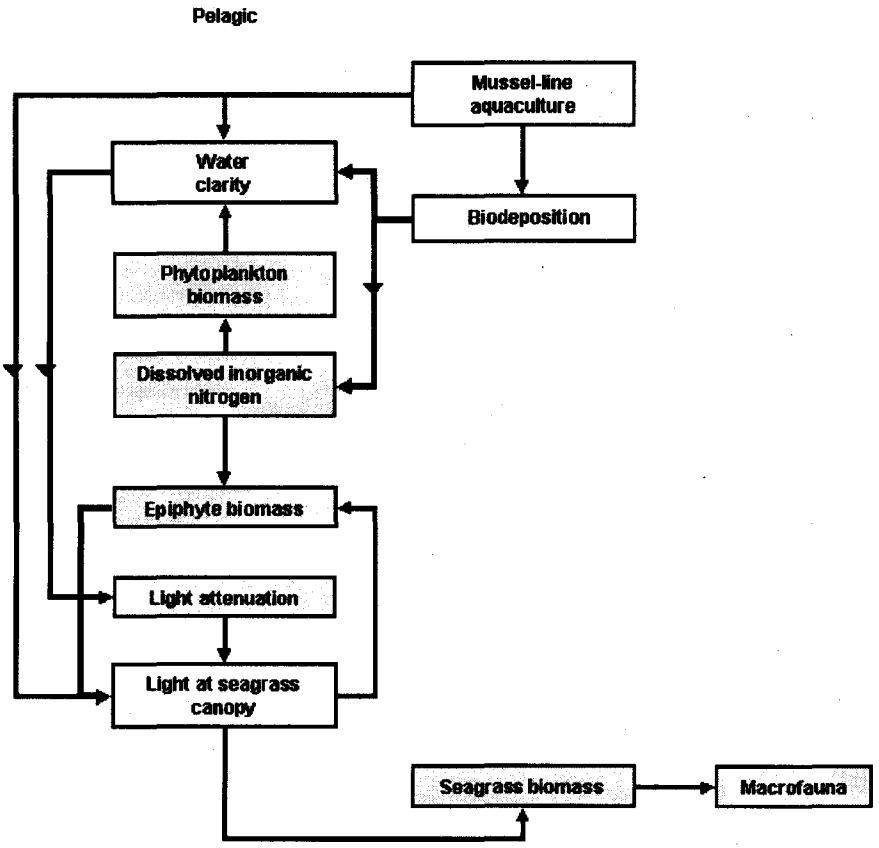


Figure 1.1 A conceptual model of the pelagic mechanisms of cause-effect from mussel-line aquaculture to seagrass communities.

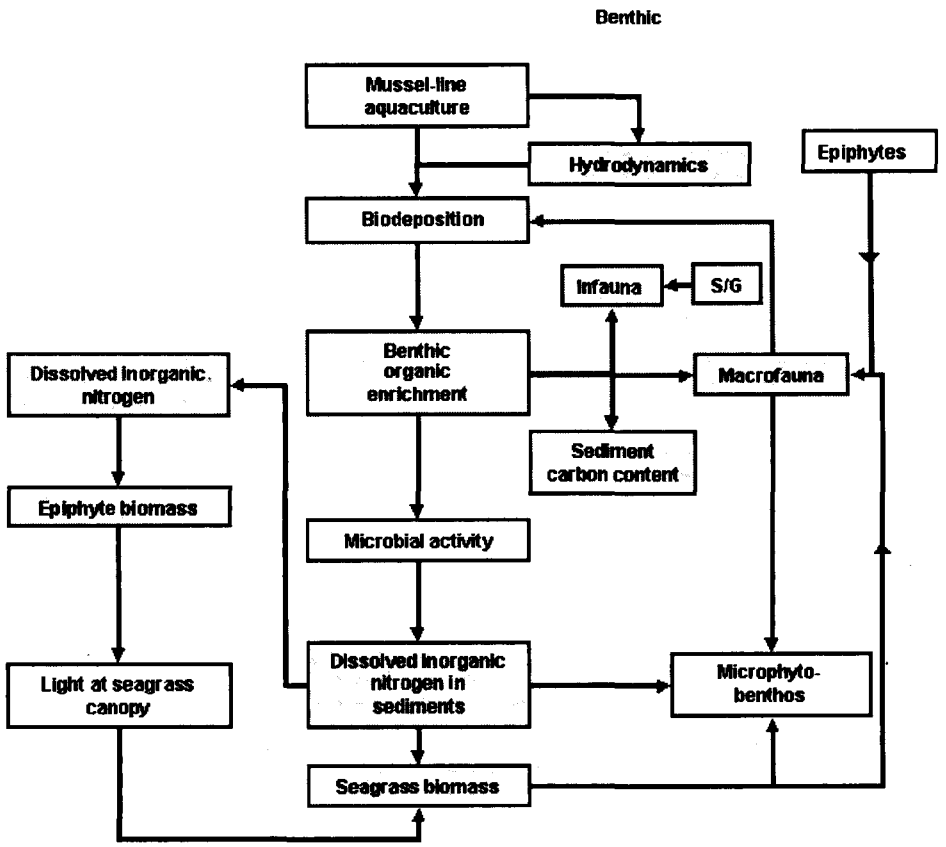


Figure 1.2 A conceptual model of the benthic mechanisms of effect from mussel-line aquaculture to seagrass communities.

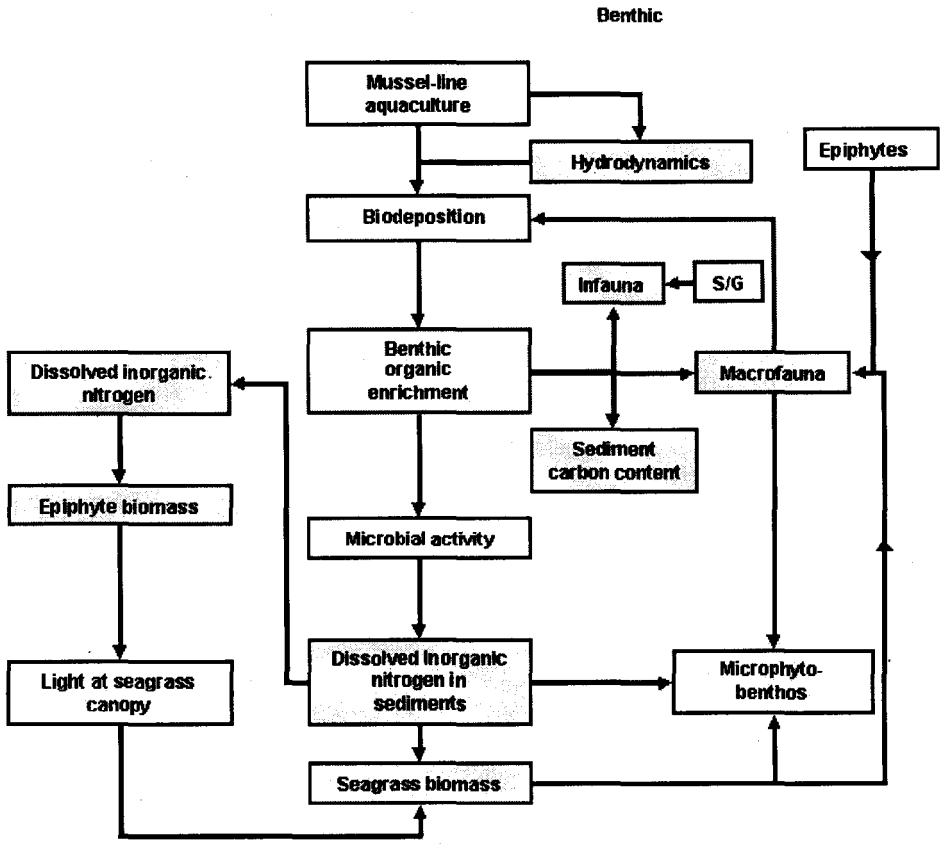


Figure 1.2 A conceptual model of the benthic mechanisms of effect from mussel-line aquaculture to seagrass communities.

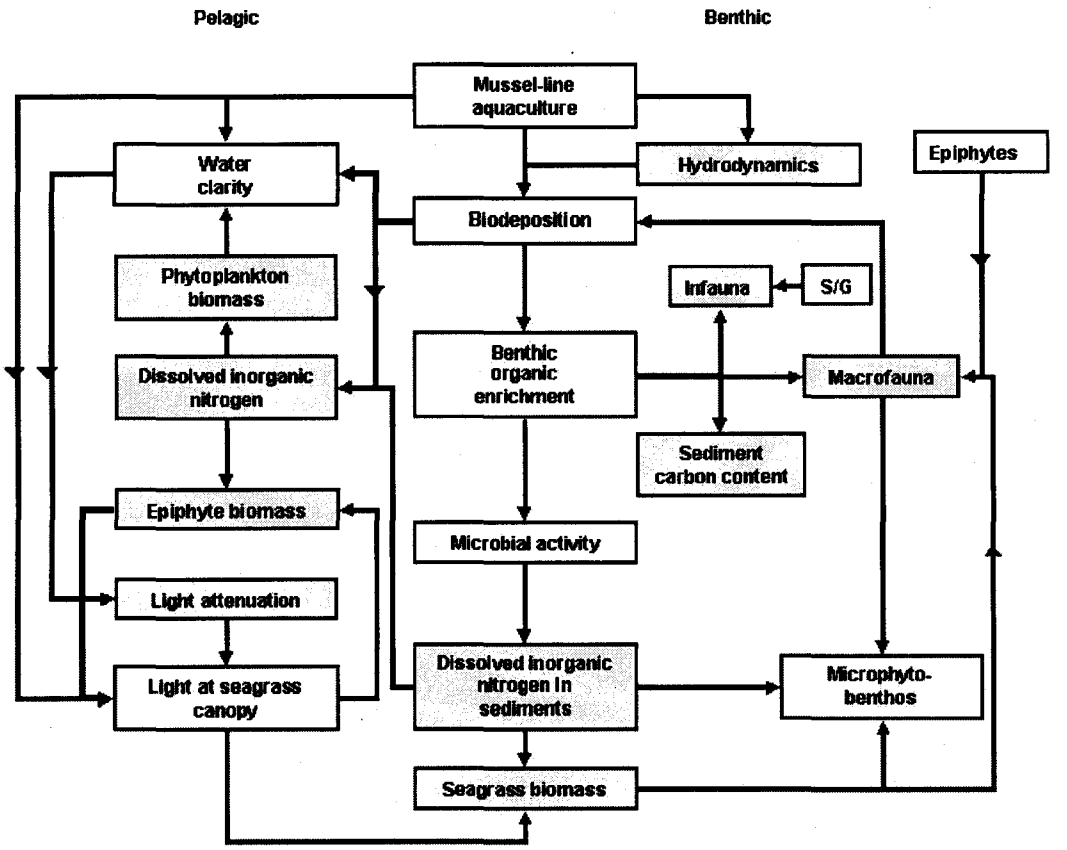


Figure 1.3 A conceptual model of combined benthic and pelagic mechanisms of effect from mussel-line aquaculture to seagrass communities.

CHAPTER 2: RESEARCH DESIGN & METHODS

2.1 THE STUDY SITE

The study site is a seagrass ecosystem located in the vicinity of the Misery Beach mussel aquaculture lease in King George Sound, Albany, Western Australia. See Appendix, Plate 1.

Figure 2.1 A map of Albany, showing the location of the study site. Modified from Image Quest Original.

In Western Australia, long-line cultures are the most common method of producing mussels (Beasley & Maguire 2000). A cable is stretched horizontally along the surface of the water, retained by using large buoys, and anchored with weighty concrete blocks. Vertical ropes hang from the main cable and hold the mussel cultures at least 4 metres off the bottom. Harvesting of mussels from long-lines involves removing the mussel cultures and putting them through a washer-tumbler machine that rotates, removing fouling organisms as the mussels rub past each other (Beasley & Maguire 2000). The washer-tumbler machines are usually operated on site and the removed fouling material is discarded overboard. This practice potentially introduces organic loads to the benthos during each harvest.

In 2000, the Misery Beach mussel aquaculture lease in King George Sound was the largest and most well established example of aquaculture located over seagrass meadows in Western Australia, and provided a valuable opportunity for this study (Figure 2.1). The Misery Beach mussel-lines were carrying approximately 21 tonnes of mussels spaced in the region of 16 hectares of *Posidonia sinuosa* meadows at a mean depth of 10 m. The lease taken by Ocean Foods International allowed for the mussel farm of approximately 25 hectares at Misery Beach in 1994. Within King George Sound, the mussel farm is protected from the Southern Ocean, but is well flushed by currents. Originally, the mussel-lines were placed perpendicular to the coastline, carrying between 20 and 60 tonnes of mussels. In 1999, storms caused the mussel-lines to shift and become bunched, but by 2000, the mussel-lines were re-set horizontal to the coastline.

Hydrodynamics are a major factor that influences the carrying capacity of the environment and the impacts of mussel aquaculture, because water current direction and velocity determine the magnitude of the dispersal of faecal material (Gibbs et al. 1992, Hatcher et al. 1994, La Rosa et al. 2000, and Chamberlain et al. 2001). Many studies suggest that the impacts of shellfish aquaculture are negligible in well-flushed ecosystems. Most impacts from mussel aquaculture have been reported from farms with high stocking densities in environments with low-carrying capacities, where water movement is low (Mattsson & Linden 1983, Kautsky & Evans 1987, Mirto et al. 2000, Chamberlain et al. 2001). In those studies, biodeposition has been reported close to the farm, and Chamberlain (2001) documented that a reduction in the density of infauna was restricted to 40m from a farm. With increasing flushing, the potential for a pelagic cause-effect mechanism is reduced, as nutrients, organic matter and phytoplankton concentrations are dispersed. Nevertheless, well-flushed ecosystems could be more prone to a benthic cause-effect mechanism (Pearson & Rosenburg 1978), through faecal material accumulating at the benthos and being incorporated into the sediment. Potentially, a benthic effect may be observed, while the water column remains unaffected.

2.2 METHODS

Sampling at Misery Beach occurred in late summer and early spring of 2000, representing the start and end of the productive period of the annual cycle (Alcoverro et al. 1995, Marba et al. 1996). During these sampling occasions, changes in ecosystem productivity were expected to be most pronounced. From the centre of the mussel aquaculture at Misery Beach, two transects were established, one running with, and the other against, the prevailing currents (Figure 2.2). Seagrass, sediments and water column parameters were randomly sampled at the centre of the farm, and then at 3 sites along each transect, at the edge of the farm, 100 m and 500 m from the

edge of the farm (Figure 2.2). The sites 500 m from the edge were considered as reference sites. A star picket marked the centre and 20m radius defined the boundaries, of each site. Parameters in the water column, seagrass and sediment were sampled at each site (Table 2.1) to assess conceptual pathways of cause and effect from mussel aquaculture to seagrass ecosystem components. The study observed parameters at sites of varying distances from a mussel aquaculture, to assess whether differences were consistent with known effects of organic enrichment, physical destruction and light depletion. When diving was required to collect samples, SCUBA was used.

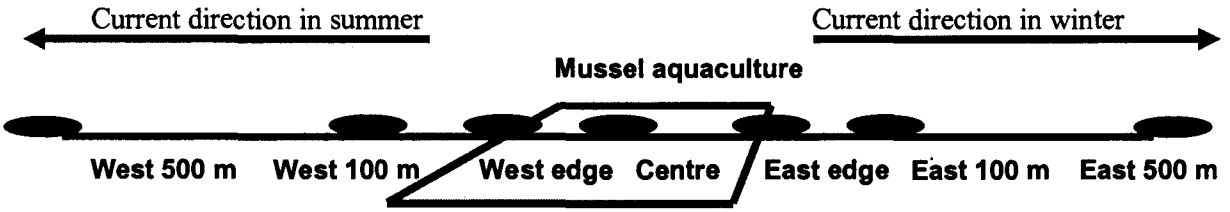


FIGURE 2.2 Location of sites along two transects running west and east of a low-density mussel aquaculture.

TABLE 2.1 Sediment, seagrass and water column parameters sampled at seven sites running along two transects through a low-density mussel aquaculture. Note: n = number of replicates.

Sediment	Seagrass	Water Column
Abundance of benthic macroinfauna (n=5)	Leaf extension (n=5)	Kjeldahl nitrogen (n=1)
Taxa richness of benthic macroinfauna (n=5)	Shoot density (n=5)	Dissolved inorganic nitrogen (n=1)
Sediment loss on ignition (n=5)	Maximum height (n=5)	Ammonium (n=1)
Chlorophyll-a (n=5)	Biomass (n=5)	Chlorophyll-a (n=1)
Porewater dissolved inorganic nitrogen (n=5)	Leaf area index (n=5)	Photosynthetically active radiation (n=1)
Porewater ammonium (n=5)	Epiphyte biomass, ashfree biomass and $CaCO_3$ biomass (n=5)	Current velocity (n=2)
Depth to the sediment (n=5)	Abundance of motile epibenthic macrofauna (n=5)	Current direction (n=2)
	Taxa richness of motile epibenthic macrofauna (n=5)	Temperature (n=1)
		Dissolved oxygen (n=1)
		pH (n=1)

2.2.1 Sediment

2.2.1.1 Porewater Dissolved Inorganic Nitrogen (DIN)

Potentially, DIN in the porewater is directly available to seagrass growing in the sediment (Reusch et al. 1994, Peterson & Heck Jr 2001a), and is possibly available to associated epiphytes through efflux (Baudinet 1990, Asmus & Asmus 1991, Asmus et al. 1994). A stainless steel sipper with an inner 20μm screen was inserted into the sediment to a depth of 10cm to collect porewater samples. Five replicate samples were taken. The samples were filtered through GF/F 0.7μm filters to remove suspended particulates. The Automated Cadmium Reduction Method was employed to determine nitrate/nitrite concentrations in filtered water samples, using a Skalar Auto Analyser. Ammonia concentrations in filtered water samples were analysed using the Phenate Method (Clesceri et al. 1998) using a Skalar Auto Analyser. DIN concentrations were calculated by adding the ammonium concentration to the nitrate/nitrite concentrations.

2.2.1.2 Sediment chlorophyll-a and loss on ignition

Chlorophyll-a in the sediment is usually associated with the microphytobenthos (Sundback 1991, MacIntyre et al. 1996, Barranguet 1997). In this study, chlorophyll-a concentrations were used to represent approximate levels of microphytobenthos. Microphytobenthos is a potential food source for benthic fauna, and is potentially influenced by the farm via increased sedimentation of microphytoplankton. Five replicate sediment samples (160ml) were collected in small acrylic cores (4.5cm in diameter and 30cm in length) to a depth of 10cm. For each, the sediment was mixed into a homogeneous slurry and a sub-sample (5ml) was taken for chlorophyll-a analysis. Chlorophyll-a was extracted from five replicate sub-samples of each using ethanol, followed by centrifugation. Chlorophyll-a was measured using a Shimazu Scientific UV mini 1240 spectrophotometer according to the methods of Jepersen and Christoffersen (1987).

Loss on ignition (LOI) represents a large proportion of organic material in the sediment, and therefore was used to indicate the amount of biodeposition at a site (Hargrave et al. 1997). From the original samples five replicate sub-samples (100 ml) were dried at 80° C for 24 hours, before combustion at 550° C for 2 hours to determine LOI.

2.2.2 Seagrass

Posidonia sinuosa meadows dominated the benthos at Misery Beach. The study took the approach of Downing & Anderson (1985), who recommended using small quadrats with a large number of replicates to sample seagrass. Where applicable, the smallest effective quadrat size was used to allow for the maximum replication that time and budget constraints would allow. Five replicate samples were taken from each site, for all seagrass and epiphytes variables except seagrass percentage cover.

2.2.2.1 Leaf extension rates and productivity

Seagrass productivity has been shown to be reduced where light conditions are unfavourably low (Silberstein et al. 1986, Gordon et al. 1994, Kuo et al. 1996). Conversely, seagrass productivity is known to increase where levels of porewater nutrients are favourably increased (Reusch et al. 1994, Peterson & Heck Jr 1999, 2001a, b). Leaf extension rates were determined using the hole-punch method of Kirkman and Reid (1979). Twenty-five shoots were punched in each of five randomly-located quadrats. Shoots were retrieved after 26 days and extension rates determined. Productivity was estimated by determining the mean increase in dry weight per leaf, multiplied by the number of leaves in each quadrat (Short & Coles 2001).

2.2.2.2 Shoot density, maximum height of seagrass, leaf area index and above ground biomass.

Seagrass shoot density and maximum height were used as indications of the seagrass health (Short 1986, Fitzpactrick 1995, Ceccherelli 1997, Delgado et al. 1999, Wood & Lavery 2000, Ruiz et al. 2001), and also as a measure of its importance as habitat (Orth et al. 1984, Bell & Westoby 1986, Edgar & Robertson 1992, Jernakoff & Nielsen 1997a, Hovel et al. 2002). At each site, the number of shoots and the maximum height of the leaves within randomly-located 20cm x 20cm ($0.04m^2$) quadrats were recorded (Short & Coles 2001). Seagrass above-ground biomass is known to be affected by organic enrichment of the benthos (Mendez et al. 1997, Delgado et al. 1999, Miller & Sluka 1999, Pergent et al. 1999, Ruiz et al. 2001, Garcia et al. 2002), and also light depletion (Fitzpatrick 1995). All above-ground material was taken from the quadrat. In the laboratory, epiphytes were scraped from the above-ground material in each replicate. Leaf area index provided a relative representation of the space available for epiphytic material at each site. Twenty-five shoots were randomly selected from each sample and the widths and length of leaves measured. The average one-sided leaf area was determined and multiplied by number of leaves per m^2 to calculate the leaf area index (m^2 of leaf per m^2) at each

site (Short & Coles 2001).

2.2.2.3 Epiphyte biomass

Epiphyte biomass is known to respond to increased nutrient levels in the lower water column, and can reduce the capacity of the seagrass to carry out photosynthesis (Silberstein et al. 1986, Frankovich 1997). Epiphytic algae is a potential food source to benthic macrofauna (Bell & Westoby 1987, Jernakoff & Nielsen 1997a, Cruz-Rivera & Hay 2000, Ruiz et al. 2001). Total dry epiphytes, ashfree epiphytes and epiphyte carbonate, can respond independently to environmental changes, and for that reason these three levels of epiphyte biomass were measured. Total epiphyte biomass was considered important measure of habitat value of epiphytes to macrofauna (Jernakoff & Nielsen 1997a), whereas ashfree epiphyte biomass is more indicative of food provision to benthic macrofauna than the other measures of epiphyte biomass (Peterson & Heck Jr 2001a, & b). Carbonate biomass can be indicative of the palatability of epiphytes, and may influence grain size of the sediments (Cambridge & Hocking 1997). Epiphyte material was retained after being scraped, using a flat blade, and dried at 80°C to constant weight to determine dry weight (Short & Coles 2001). It was then combusted at 550°C for two hours, and weighed to determine loss on ignition. Glucose standards were included with the samples exposed to 550°C for attaining measurements to correct for incomplete combustion. Samples were then combusted at 950°C for two hours, desiccated and weighed to determine calcium carbonate content. Calcium carbonate standards were included with the samples exposed to 950°C for attaining measurements to correct for incomplete combustion.

2.2.2.4 Seagrass % coverage

The boundaries of a dominant seagrass species is determined by the percentage cover relative to other species of seagrass, macroalgae and bare sediment (Kirkman & Kuo 1990). Diversity of macrofaunal communities can be influenced by the percentage cover of seagrass to the benthos (Sogard et al. 1987, Connolly & Butler 1996, Gambi et al. 1998). Six 20m long transects were set up radiating from the central point of each site. Twenty-four percent of the area of each site was recorded using video. This video was held 1m above the seagrass canopy, with a field of view covering $1m^2$ per metre of transect. The percentage cover of seagrass was calculated in the laboratory, using the video footage from each site, following the methods of Meese & Tomich (1992).

2.2.3 Benthic macrofauna

2.2.3.1 Motile epibenthic macrofauna

Motile epibenthic macrofauna are a vital link in the food web within seagrass ecosystems, and are considered to be useful indicators of changes within seagrass ecosystems (Bell & Westoby 1986, Sergeev et al. 1988, Edgar & Robertson 1992, Connolly 1995, Connolly & Butler 1996, Jernakoff & Nielsen 1997b, Sanchez-Jerez et al. 2000, Lee et al. 2001, Hovel et al. 2002). Nets comprising 100μm mesh with an opening of 25x25cm (0.0625m^2) were used to sample motile epibenthic macrofauna using the methods of Connolly (1997) and Sergeev et al. (1988). Each net was rapidly placed over the sediment surface, retaining all material larger than 100μm within the mesh bag. Without lifting the net, the seagrass was cut off using sharp sheers that were held flush with the sediment surface, before the bag was sealed with a mesh cover. Retained material was washed through two stacked sieves, a 4mm sieve that retained larger macrofauna (>4mm), and a 1mm sieve that retained small, motile epibenthic macrofauna (1<4mm). Five replicate samples were taken from each site. The samples were preserved in 5% formalin buffered in seawater and transferred to 70% ethanol for further sorting. Individuals were identified to family level and abundances of individuals in each family were recorded. While, family level identification is considered adequate for this study, it is possible that the analysis lacks a degree analytical resolution. This should be considered when interpreting the results of this study.

2.2.3.2 Benthic macroinfauna

Benthic macroinfauna assemblages are well documented as being effective indicators of organic enrichment (Warwick 1988, Weston 1990, Hutchings et al. 1991, Clarke 1993, Snelgrove & Butman 1994, Hargrave et al. 1997, Beesley et al. 2000). A stainless steel corer, 11.5cm diameter and 18cm long, was used to collect sediment samples to a depth of 15cm for benthic macroinfauna. Samples were taken to 15cm to ensure maximum representation of biomass and species richness. Each core sample was washed through two stacked sieves, a 4mm sieve that retained larger infauna (>4mm), and a 1mm sieve that retained small, infauna (1<4mm). Infauna was fixed in 5% formalin buffered in seawater and transferred to 70% ethanol for further sorting under a dissecting microscope. Five replicate samples were taken from each site. Infauna taxa were classified to family level. Warwick (1988) and Karakassis & Hatziyanni (2000) found that when taxa were identified to family rather than genus or species levels, information loss regarding macrofauna community structure is negligible. While, family level identification is considered adequate for this study, it is possible that the analysis lacks a degree analytical resolution. This should be considered when interpreting the results of this study.

2.2.4 Water column

Given the scale in variability of the water column at Misery Beach, water column variables were not replicated. Although there was no substantial benefit to be gained from additional replication of variables, it must be considered when interpreting the results of this study.

2.2.4.1 Hydrodynamics

Current velocities and directions are considered to be major factors affecting the dispersal and distribution of waste products from aquaculture (Wu 1994, Findlay & Watling 1995, Findlay 1997, Hargrave et al. 1997, Henderson et al. 2001). Two submergible drogues were placed at the surface (0-1m) and two at the bottom of the water column (6-8m) to determine current direction and velocity of surface water and deeper water, according to the methods of D'Adamo et al. (1992). Direction and distance from an initial location were plotted over time using a Global Positioning System.

2.2.4.2 General water quality

General abiotic components of the water column were measured to assess any potential dissimilarity between sites. Photosynthetic active radiation (PAR), dissolved oxygen, pH, redox potential, turbidity, temperature and salinity were measured at 1m intervals from the surface to the bottom of each site using a Yeokal Metre.

2.2.4.3 Nutrients

Mussel aquaculture is known to have the capacity to convert particulate organic nitrogen (PON) to dissolved inorganic nitrogen (DIN), which is available for uptake by phytoplankton (Asmus & Asmus 1991) and macrophytes, including seagrass and associated epiphytic algae(Fabiano et al. 1994, Reusch et al. 1994, Peterson & Heck Jr 2001a, b). At each site, water samples from 50cm below the surface of the water column and 50cm above the sediment were collected for nutrient analysis. Samples for analysis of DIN and ammonium were filtered through Whatman GF/F filters (0.7μm), while those for analysis of Kjeldahl nitrogen were unfiltered. All nutrient samples were analysed using a Skalar Auto Analyser. The Automated Cadmium Reduction Method was employed to determine DIN concentrations. Ammonium concentrations were analysed using the Phenate Method (Clesceri et al. 1998). Kjeldahl nitrogen concentrations were determined using the Automated Cadmium Reduction Method after Kjeldahl Digestion (Clesceri et al. 1998).

2.2.4.4 Chlorophyll-a

Chlorophyll-a in the water column was used to indicate the concentration of phytoplankton at each site. Three litres of water from the bottom and the top of the water column at each site were filtered through Whatman GF/C filters (1.2μm). The filters were stored on ice, in the absence of light. The filter papers were ground using a TFE/glass grinder for pigment extraction. Chlorophyll-a pigment was extracted using ethanol, and measured using a Shimazu Scientific UV mini 1240 spectrophotometer according to the methods of Jepersen and Christoffersen (1987).

2.3 DATA ANALYSIS

2.3.1 Univariate analysis

Two-way analysis of variance was used to test for differences between means from each site for the variables in the Table 2.2, using SPSS™ (SPSS INC 2002). Data were first tested for homogeneity of variance.

2.3.1.1 homogeneity of variance

As a first step, data sets were tested for homogeneity using Levene's Test within SPSS. If Levene's Test revealed that variances between sites were heterogenous, an appropriate transformation method was employed. Table 2.2 shows the type of transformation used for each parameter. Subsequently, Levene's homogeneity of variance tests on transformed data showed that variances between sites for seagrass leaf extension (mg), chlorophyll-a, the ratio of ashfree epiphyte to seagrass biomass, and macroinfauna abundance and taxa richness remained heterogeneous. However, ANOVA is considered to be robust to moderate non-homogeneity of variance when sample sizes are equal (Underwood 1997, McGuinness 2002). It was therefore appropriate to continue with parametric analysis of the data, and the possibility of an invalid conclusion was accounted for, by setting the significance level at $P < 0.01$ for variables that failed to demonstrate homogeneity of variance. The significance level was set at $P < 0.05$ for the majority of variables, where variance was homogenous between sites.

Table 2.2 Variables statistically tested for differences between means using two ANOVA, and the type of prior transformation used.

Type of Variable	Natural Log transformed	untransformed	Square root transformed
Sediment	Loss on ignition Chlorophyll-a		Porewater ammonium Sediment chlorophyll-a
Seagrass	Biomass Shoot density Leaf density Leaf area index Leaf extension (mm)	Maximum height	Leaf extension (mg)
Epiphytes	Biomass Ashfree biomass Calcium carbonate biomass Epiphyte biomass to seagrass biomass		
Motile epibenthic macrofauna	Abundance Taxa richness		
Benthic infauna	Abundance		Taxa richness

2.3.1.2 Post Hoc tests

When differences among sites were found to be significant, multiple pairwise *Post Hoc* comparisons of the means were performed using Tukey's tests to determine which pairs of means were different.

2.3.2 Multivariate analysis

2.3.2.1 Ordinations

Ordinations were generated for visual representation of the dissimilarity among epibenthic macrofauna and macroinfauna assemblages at different sites. The ordinations were generated using PRIMER's nMDS module based on Bray-Curtis dissimilarity matrices, using non-metric multidimensional scaling (nMDS) (Clarke & Warwick 1994).

2.3.2.2 Analysis of similarities (ANOSIM)

ANOSIM is a non-parametric test of the significance of the patterns generated by ordination (Clarke & Warwick 1994). Using PRIMER, each ANOSIM is based on a Bray-Curtis dissimilarity matrix, and was used to test for significant differences ($P < 0.05$) among sites. ANOSIM results were run using square root transformed data and also presence-absence data. The purpose of this is to compare the two different results; one set of results pertaining to taxa richness and abundance, the other pertaining to taxa richness only.

2.3.2.3 Simper

Exploratory analysis using the Simper module within PRIMER was used to select groups of species contributing 95% of the nMDS pattern. Simper analysis was run for epibenthic macrofauna and macroinfauna for summer and spring, on the same Bray-Curtis dissimilarity matrices that were produced for ordinations and ANOSIM. Simper helped identify which individual species accounted for the patterns in dissimilarity between sites (Clarke & Warwick 1994).

2.3.2.4 BVSTEP

Exploratory analysis using the BVSTEP module within PRIMER was used to select combinations of variables "explaining" 95% of the nMDS pattern for benthic macrofauna. The BVSTEP routine compares the dissimilarity matrix for the benthic macrofaunal data with the dissimilarity matrix for the other environmental data. A step-wise routine searches through potential combinations of environmental variables that best match the patterns of macrofaunal assemblages (nMDS) (Clarke & Warwick 1994).

CHAPTER 3: RESULTS

3.1 WATER COLUMN

3.1.1 Hydrodynamics

Coastal water currents at Misery Beach were recorded at a speed of 0.136 m s^{-1} at a direction of 270° (westerly) in January, 0.061 m s^{-1} at 168° (south easterly) in May and 0.075 m s^{-1} at 83° (easterly) in October. The current speeds and direction recorded in this study correspond with data for Albany's prevailing winds during those months (Meteorology 1999). This supports the general rule that current speeds and directions in shallow coastal areas are controlled by local wind speeds and directions (D'Adamo per. com.). These findings also confirm qualitative observations of strong westward currents during summer and moderate eastward currents during spring. The mean depth of the 7 sites was 10m, and the mean difference in depths between sites was 1.6 meters (Figure 3.1). The greatest depth difference between sites was 3.4 meters, between the West 100m and the East 500m sites.

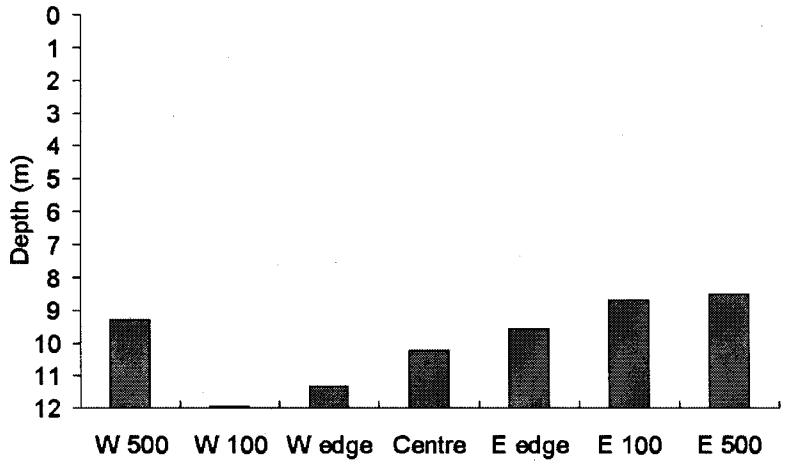


Figure 3.1 Depth of the benthos from the surface of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 5).

3.1.2 Water quality

There are no replicate data for the water column variables, nutrient concentrations, chlorophyll-a, and light. These variables show no clear trends in either summer or spring among sites along a transect running through the mussel farm (Figures 3.2 to 3.12).

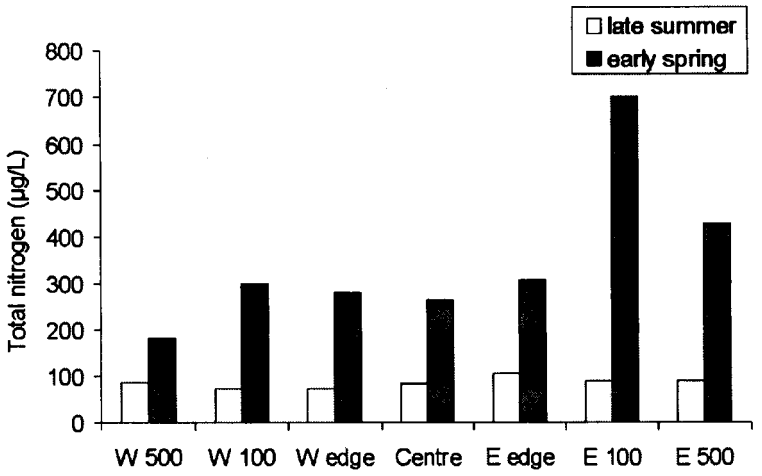


Figure 3.2 Total nitrogen concentrations in the top of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

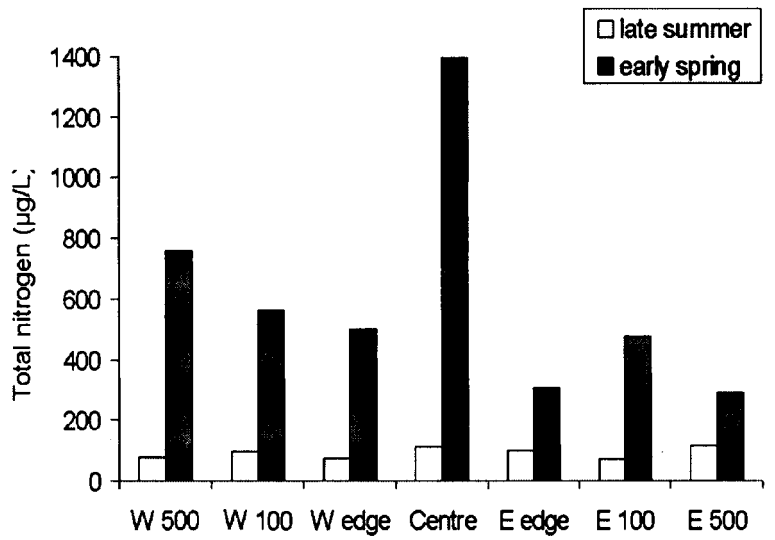


Figure 3.3 (a) Total nitrogen concentrations in the bottom of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

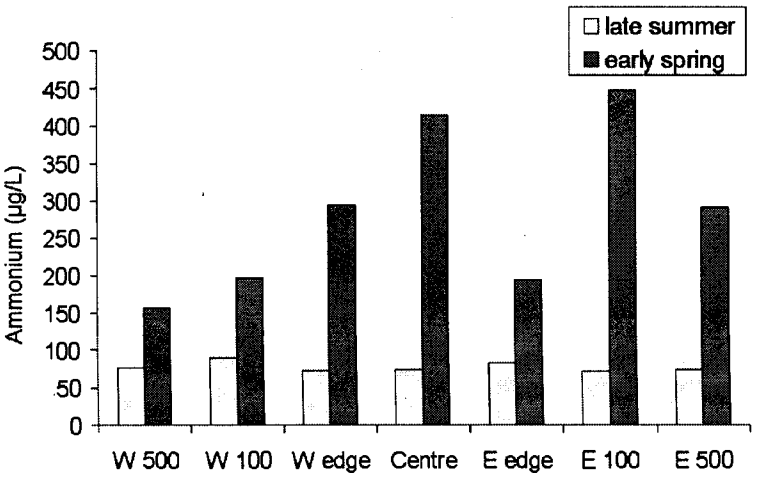


Figure 3.4 Ammonium concentrations in the top of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

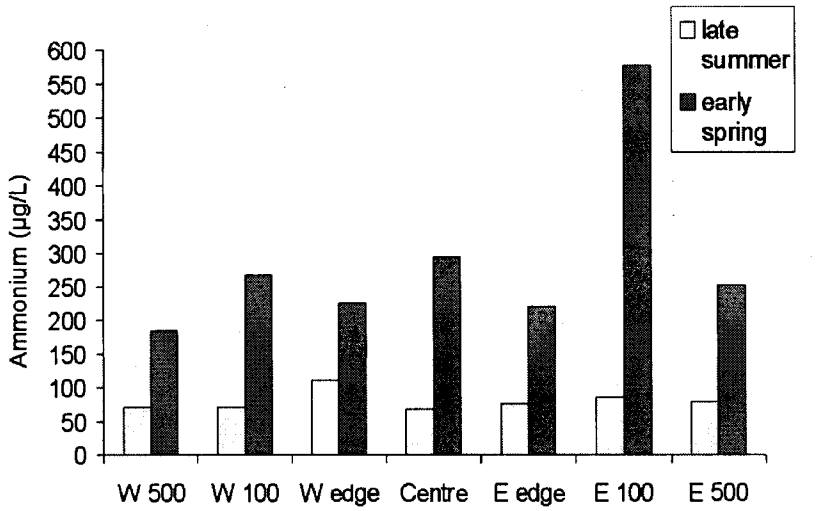


Figure 3.5 Ammonium concentrations in the bottom of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

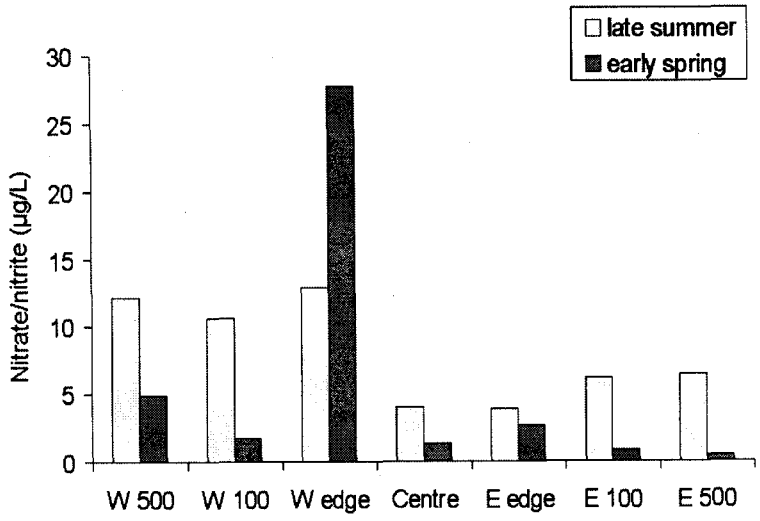


Figure 3.6 Nitrate/nitrite concentrations in the top of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

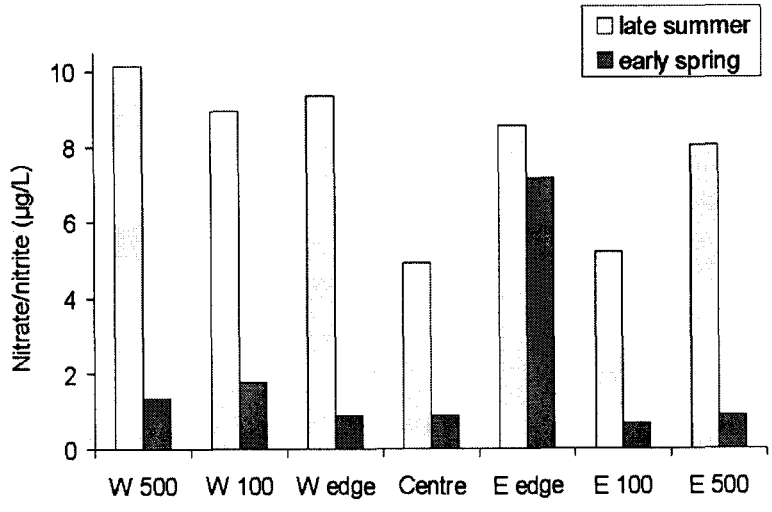


Figure 3.7 Nitrate/nitrite concentrations in the bottom of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

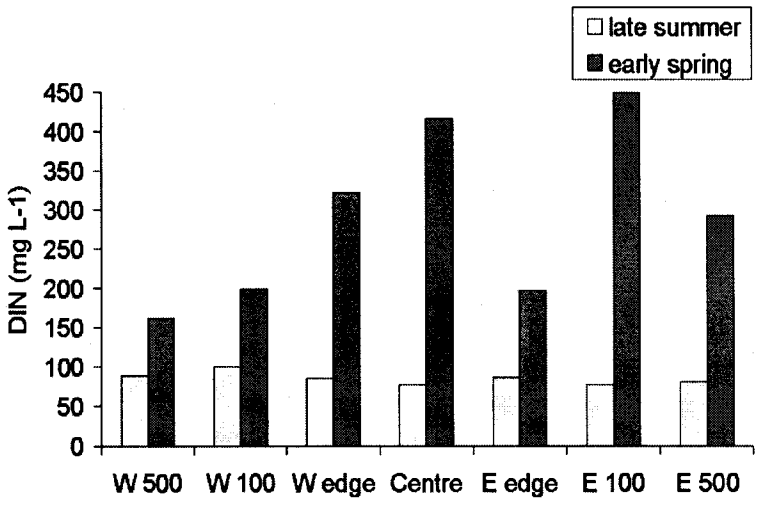


Figure 3.8 Dissolved inorganic nitrogen concentrations in the top of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

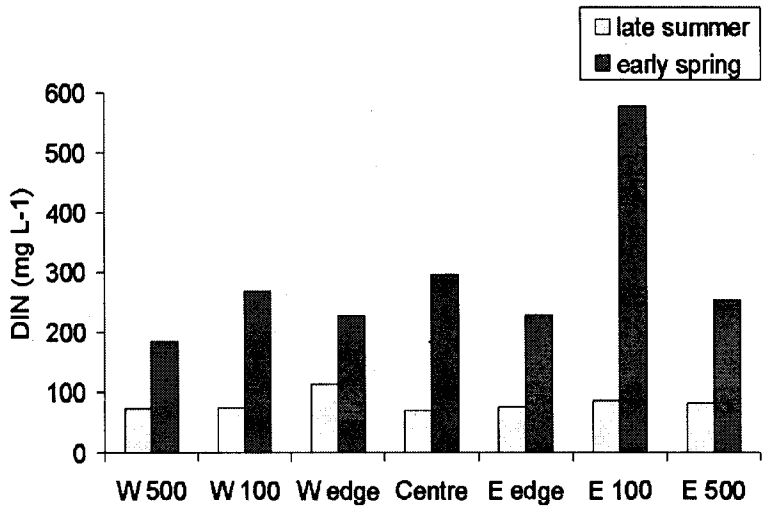


Figure 3.9 Dissolved inorganic nitrogen concentrations in the bottom of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

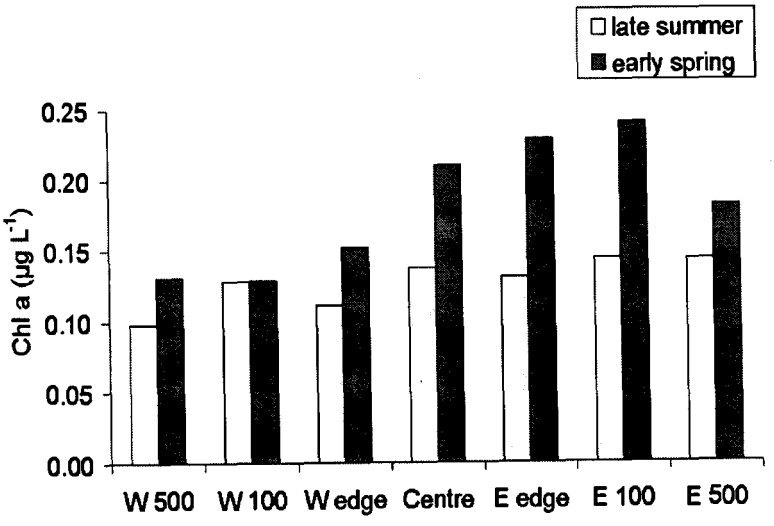


Figure 3.10 Chlorophyll-a concentrations in the top of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

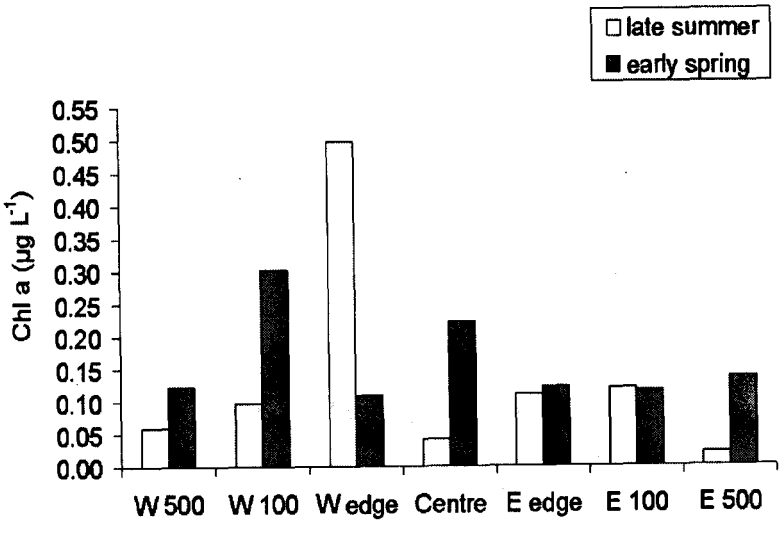


Figure 3.11 Chlorophyll-a concentrations in the bottom of the water column, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

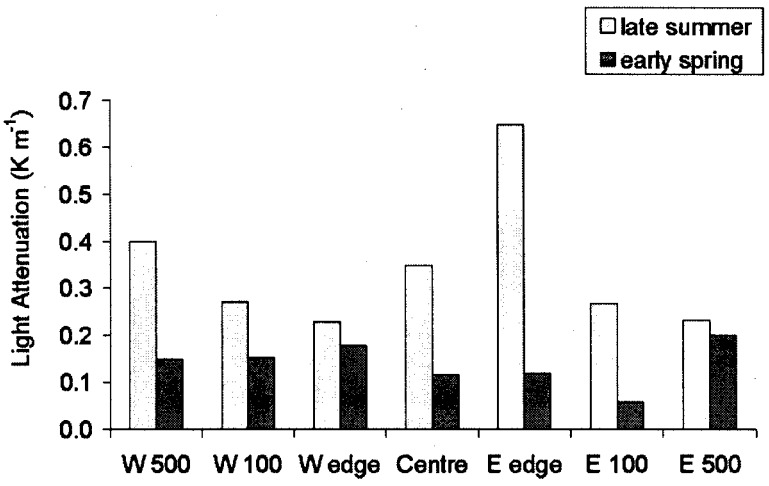


Figure 3.12 Light attenuation from the surface of the water column to the benthos, at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 1).

3.2 SEDIMENT

There is a significant site x season interaction for ammonium concentrations in the porewater (Table 3.1). Tukey's tests show that in summer the porewater ammonium concentrations were significantly lower at the West and East 500 m sites than at the West and East edge sites, whereas in spring there were no significant differences among sites (Figure 3.13b). While ANOVA shows that there were no significant differences among sites or seasons for loss on ignition (LOI) in the sediment, it shows a significant site x season interaction (Table 3.1). Tukey's tests show that LOI at the West 500 m site was significantly lower than all other sites in spring (Figure 3.14c). ANOVA showed that there were no significant differences in sediment chlorophyll-a among sites, but there was a significant seasonal difference in sediment chlorophyll-a, which was highest west of the farm in summer and east of the farm in spring (Table 3.1, Figure 3.15).

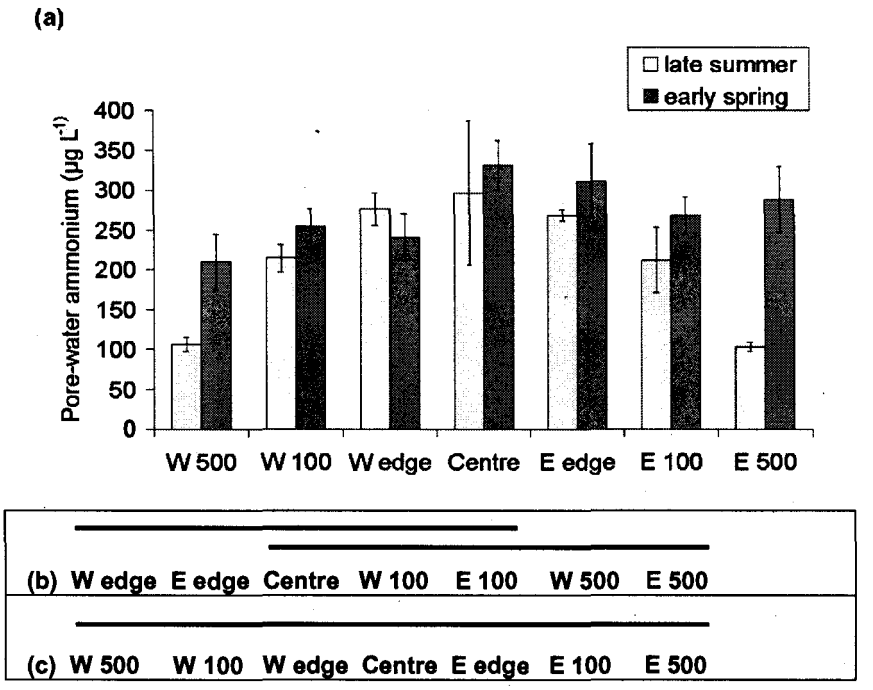


Figure 3.13 (a) Mean porewater ammonium (± SE) at sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5). **(b)** Tukey's pairwise comparisons of porewater ammonium in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of porewater ammonium at sites in spring ($p < 0.05$).

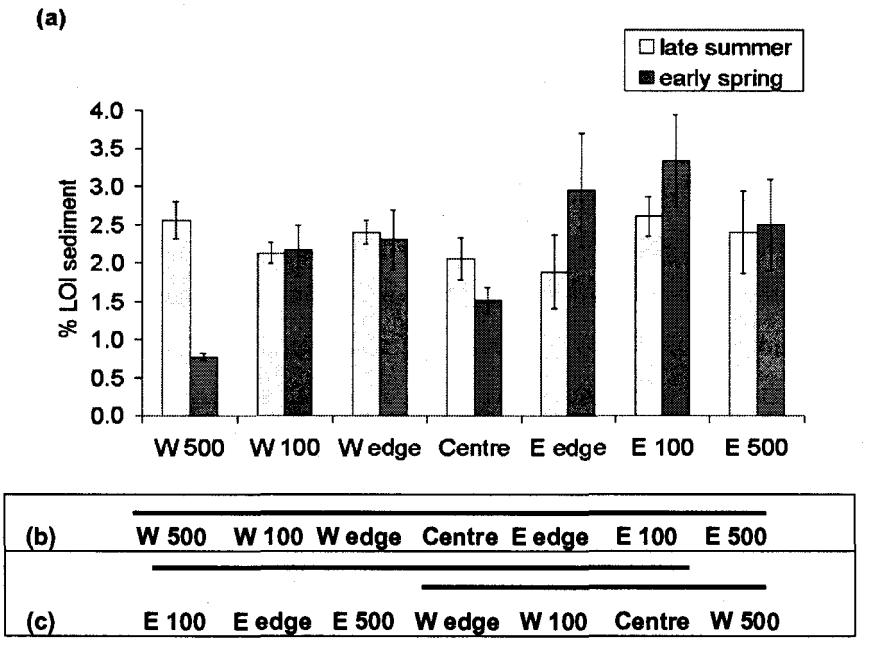


Figure 3.14 (a) Mean loss on ignition from the sediment (± SE) at sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5). **(b)** Tukey's pairwise comparisons of loss of ignition from the sediment in summer ($p < 0.05$). **(c)** Tukey's pairwise comparisons of epibenthic macrofauna abundance at sites in spring; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$).

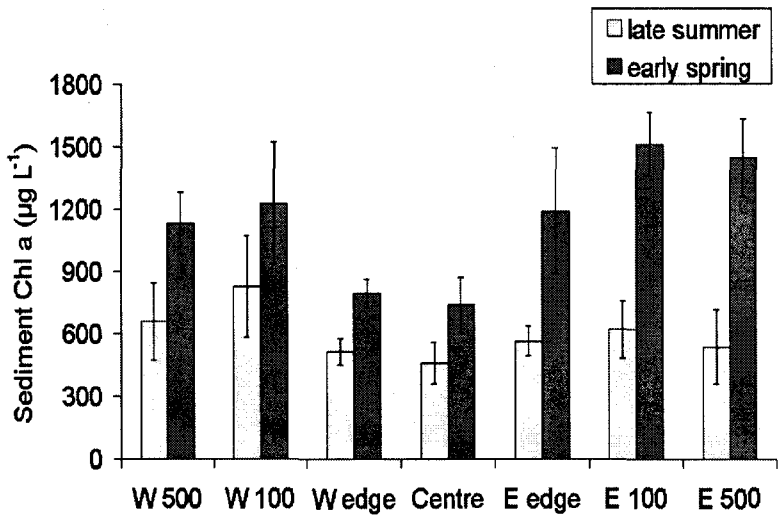


Figure 3.15 Mean sediment chlorophyll-a (± SE), at sites along a transect running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5).

Table 3.1 Two-way ANOVA results of sediment data from seven sites along transects, running west and east from a low-density mussel-line aquaculture farm.

Variables	source of variation	df	MS	F ratio	P
Ammonium in porewater	Site	6	30.725	1.603	0.290
	Season	1	78.456	4.093	0.090
	Site x Season	6	19.167	5.216	0.001
	Residual	56	3.674		
Loss on ignition of sediments	Site	6	0.525	0.771	0.620
	Season	1	0.619	0.909	0.377
	Site x Season	6	0.680	5.079	0.001
	Residual	56	0.134		
Chlorophyll-a in sediments	Site	6	0.164	0.565	0.748
	Season	1	7.543	25.978	0.002
	Site x Season	6	0.290	1.745	0.127
	Residual	56	0.166		

3.3 SEAGRASS AND EPIPHYTES

There are significant site x season interactions for parameters; seagrass biomass, leaf density, shoot density, leaf area index (Table 3.2), epiphyte biomass and epiphyte $CaCO_3$ biomass (Table 3.3). Due to these interactions, trends among sites have been examined for each season.

Seagrass biomasses, leaf densities and leaf area indices displayed a trend during summer, whereby values at the East 500 m site are shown by Tukey's test to be significantly lower than at all other sites (Fig. 3.16b, 3.18b & 3.19b). ANOVA also shows a seasonal difference in seagrass maximum leaf heights and also leaf extension rates (Table 3.2, Fig. 3.20 & 3.21).

Epiphyte biomasses and epiphyte $CaCO_3$ biomasses display trends similar to those seen for the above seagrass variables. Values at the West and East, 100 m and Edge sites and Centre site are shown by Tukey's tests to be significantly higher than at the West and East 500 m sites during summer (Fig. 3.22b & 3.24b).

In comparison to summer, during spring seagrass biomasses, leaf area indices, epiphyte biomasses, epiphyte ashfree biomasses, epiphyte calcium carbonate were significantly higher at the East 500 m site than the West 100 m site (Fig. 3.16c, 3.19c, 3.22c, 3.23c & 3.24c). Epiphyte biomasses, ashfree epiphyte biomasses and epiphyte $CaCO_3$ at the East 500 m site were also significantly higher than at the West and East edge sites (Table 3.3, Fig. 3.22, 3.23c & 3.24c) and the East 100 m in relation to ashfree epiphyte biomass (Table 3.3, Fig. 3.23c). During spring, leaf densities at the Centre site and East 500 m sites were significantly higher than at the East 100 m site (Table 3.2, Fig. 3.18c).

Tukey's test showed that unlike other seagrass and epiphyte variables in summer, shoot density was significantly higher at the West 500 m and East 500 m sites than other sites (Table 3.2, Fig. 3.17b). Of the results for seagrass, the trend for shoot densities is the most consistent with the proximity of the research sites to the mussel farm. Additionally, shoot densities at the East edge site were significantly higher than at the West 100 m site. Similar to summer, but unlike other seagrass and epiphyte variables during spring, shoot densities at the East 500 m site were significantly higher than at the Centre site (Fig 3.18c).

Table 3.2 Two-way ANOVA results for seagrass variables from seven sites along transects, running west and east from a low-density mussel-line aquaculture farm.

Variables	source of variation	df	MS	F ratio	P
Seagrass biomass	Site	6	0.666	0.239	0.947
	Season	1	11.572	4.158	0.088
	Site x Season	6	2.783	13.954	0.001
	Residual	56	0.199		
Seagrass shoot density	Site	6	0.355	0.531	0.770
	Season	1	0.224	0.336	0.583
	Site x Season	6	0.668	5.937	0.001
	Residual	56	0.113		
Seagrass leaf density	Site	6	1.054	0.408	0.850
	Season	1	0.020	0.008	0.932
	Site x Season	6	2.582	21.264	0.001
	Residual	56	0.121		
Seagrass leaf area index	Site	6	0.964	0.301	0.915
	Season	1	6.185	1.929	0.214
	Site x Season	6	3.206	19.386	0.001
	Residual	56	0.165		
Seagrass leaf extension (mm)	Site	6	1.442	2.302	0.165
	Season	1	16.495	26.524	0.002
	Site x Season	6	0.622	2.320	0.112
	Residual	56	0.342		
Seagrass maximum leaf height	Site	6	554.507	1.558	0.302
	Season	1	4204.000	11.809	0.014
	Site x Season	6	356.017	2.022	0.078
	Residual	56	176.102		

Table 3.3 Two-way ANOVA results for epiphytic algae variables from seven sites along transects, running west and east from a low-density mussel-line aquaculture.

Variables	source of variation	df	MS	F ratio	P
Epiphyte biomass	Site	6	1.111	0.416	0.845
	Season	1	14.183	5.309	0.061
	Site x Season	6	2.671	12.822	0.001
	Residual	56	0.208		
Epiphyte ashfree biomass	Site	6	0.486	0.243	0.945
	Season	1	19.503	9.774	0.020
	Site x Season	6	1.995	3.543	0.005
	Residual	56	0.563		
Epiphyte calcium carbonate biomass	Site	6	1.103	0.364	0.878
	Season	1	32.724	10.800	0.017
	Site x Season	6	3.030	15.620	0.001
	Residual	56	0.194		

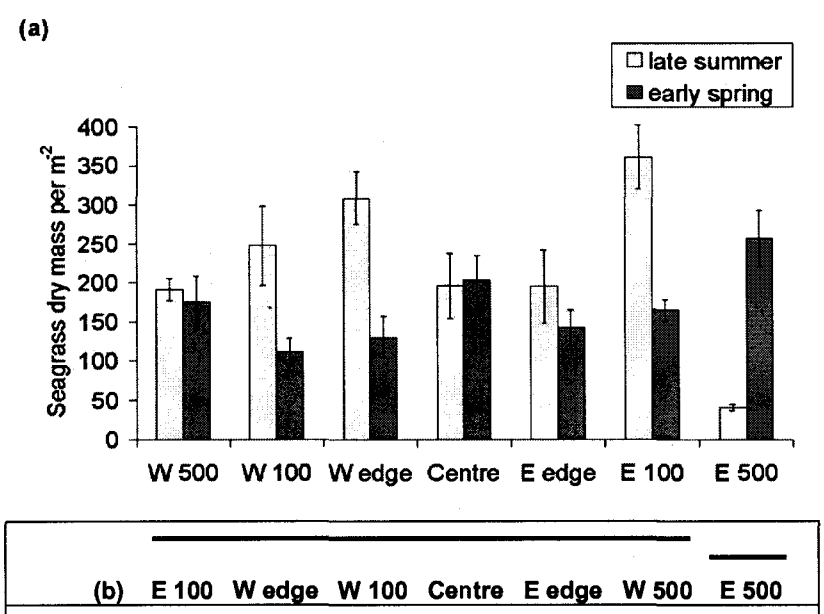


Figure 3.16 (a) Mean seagrass biomass (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5). **(b)** Tukey's pairwise comparisons of seagrass biomass at sites in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of seagrass biomass at sites in spring ($p < 0.05$).

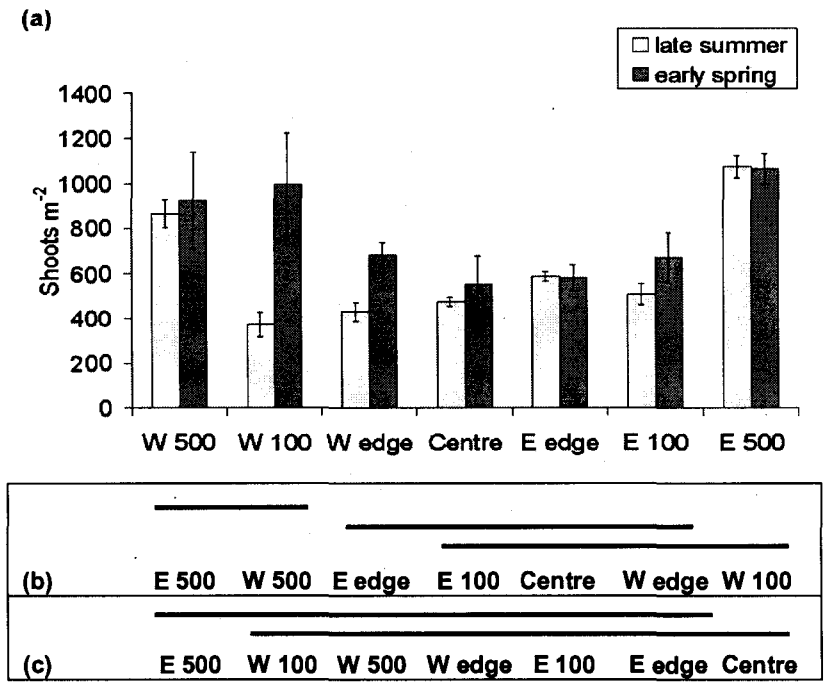


Figure 3.17 (a) Mean seagrass shoot densities (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5).**(b)** Tukey's pairwise comparisons of shoot densities in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$).**(c)** Tukey's pairwise comparisons of shoot densities at sites in spring ($p < 0.05$).

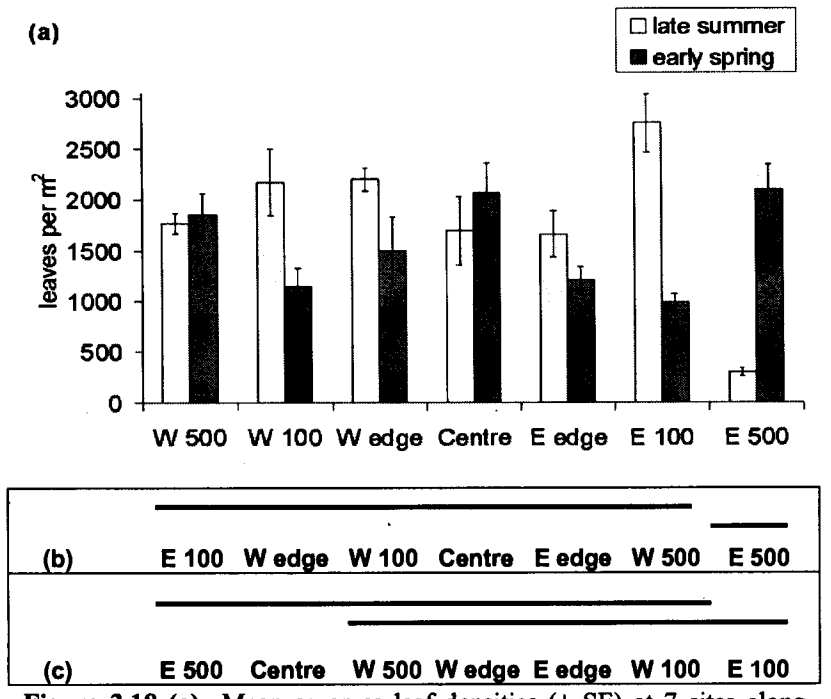


Figure 3.18 (a) Mean seagrass leaf densities (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5). **(b)** Tukey's pairwise comparisons of leaf densities at sites in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of leaf densities at sites in spring ($p < 0.05$).

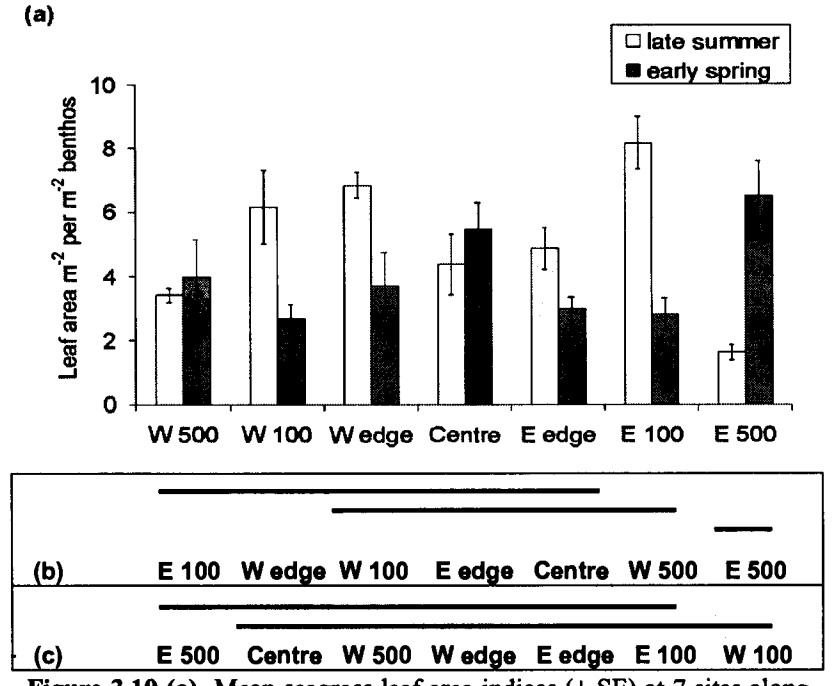


Figure 3.19 (a) Mean seagrass leaf area indices (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5). **(b)** Tukey's pairwise comparisons of leaf area indices in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of leaf area indices at sites in spring ($p < 0.05$).

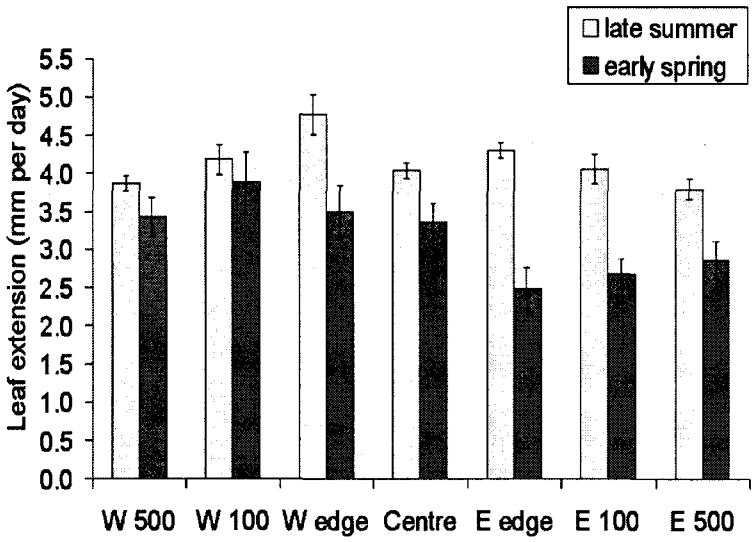


Figure 3.20 Mean seagrass leaf extension (mm/day, ± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5).

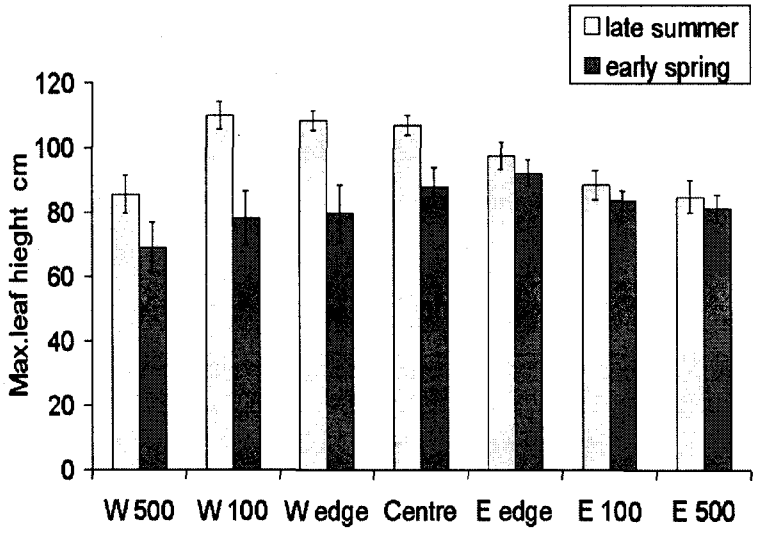


Figure 3.21 Mean seagrass maximum leaf height (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5).

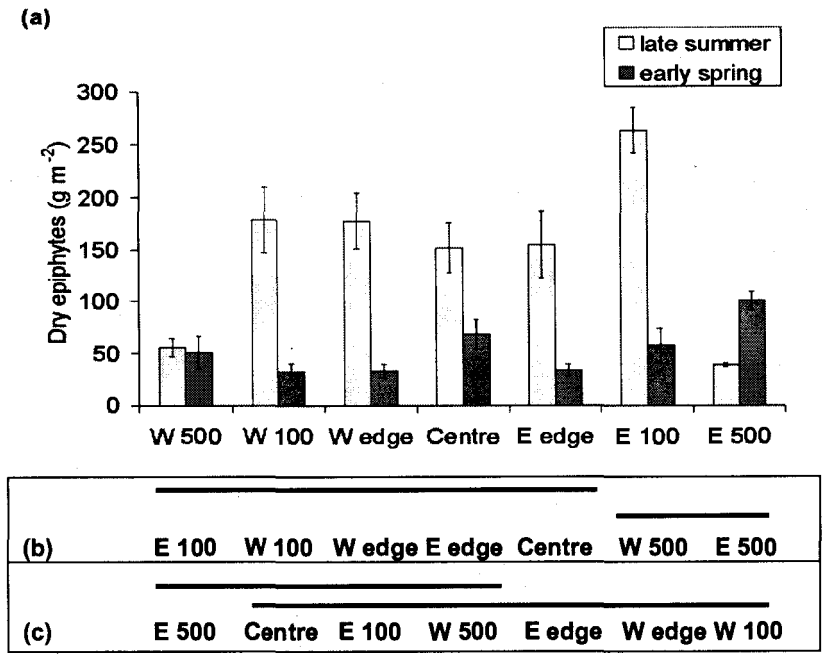


Figure 3.22 (a) Mean epiphyte biomass (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach, 2000 (n = 5). **(b)** Tukey's pairwise comparisons of epiphyte biomass at sites in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of epiphyte biomass at sites in spring ($p < 0.05$).

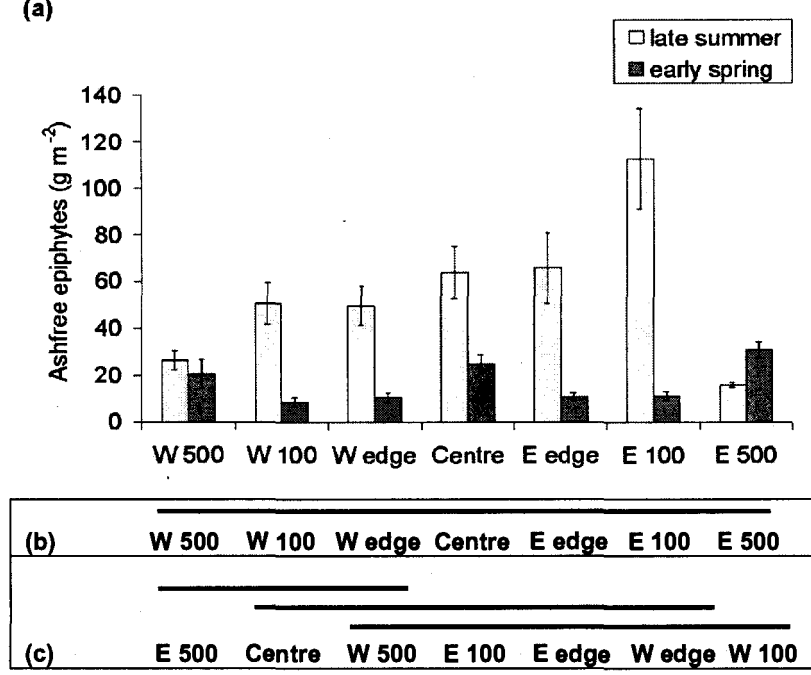


Figure 3.23 (a) Mean epiphyte ashfree biomass (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5). **(b)** Tukey's pairwise comparisons of ashfree biomass at sites in summer ($p < 0.05$). **(c)** Tukey's pairwise comparisons of ashfree biomass at sites in spring; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$).

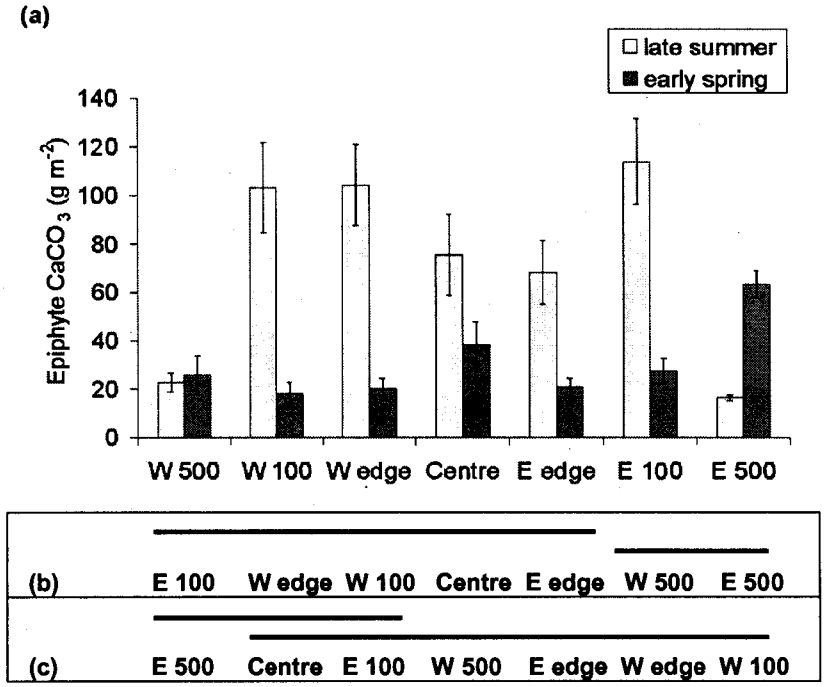
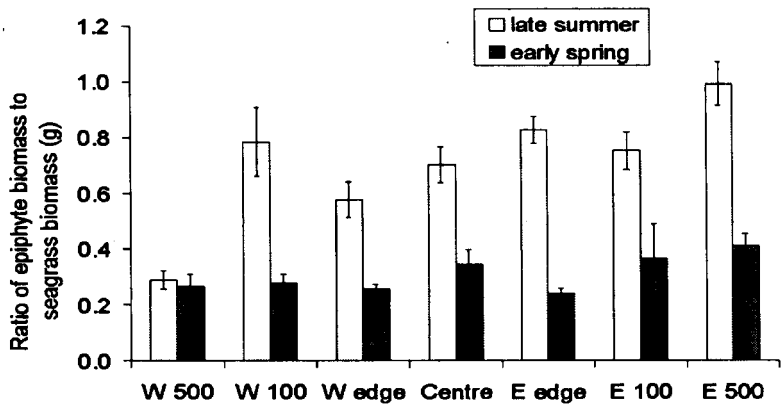


Figure 3.24 (a) Mean epiphyte calcium carbonate biomass (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5). **(b)** Tukey's pairwise comparisons of calcium carbonate biomass at sites in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of calcium carbonate biomass at sites in spring ($p < 0.05$).

ANOVA showed that there was a significant site x season interaction in ratios of dry epiphyte to seagrass biomass (Table 3.4). Tukey's tests showed that the ratio of dry epiphyte to seagrass biomass in summer was significantly lower at the West 500 site than other sites, except the West edge site (Figure 3.25b). However, there were no significant differences in the ratios of dry epiphyte to seagrass biomass among sites in spring (Figure 3.25c).

Table 3.4 Two-way ANOVA results of seagrass and epiphyte ratio data, from seven sites along transects, running west and east from a low-density mussel-line aquaculture farm.

Variables	source of variation	df	MS	F ratio	P
Ratio of epiphyte biomass to seagrass biomass	Site	6	0.170	1.727	0.262
	Season	1	2.725	27.692	0.002
	Site x Season	6	0.098	4.685	0.001
	Residual	56	0.021		



(b)	E 500	E edge	E 100	Centre	W 100	W edge	W 500
(c)	W 500	W 100	W edge	Centre	E edge	E 100	E 500

Figure 3.25 (a) Ratios of epiphyte biomass to *Posidonia sinuosa* biomass (± SE) at 7 sites along transects running west and east from a low-density mussel-line aquaculture at Misery Beach (n = 5).
(b) Tukey's pairwise comparisons of the ratios of epiphyte biomass to seagrass biomass, at sites in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of ratios of epiphyte biomass to seagrass biomass, at sites in spring ($p < 0.05$).

3.4 MOTILE EPIBENTHIC MACROFAUNA: TAXA RICHNESS AND DENSITIES

ANOVA showed a significant site x season interaction for epibenthic faunal abundance, but no significant main effect (Table 3.5). Due to this interaction, Tukey's tests have been used to examine trends among sites within each season. Tukey's test showed that, in summer, abundance of epibenthic macrofauna was significantly greater at the West 100 m site than at the West edge and Centre sites (Figure 3.26). In spring, epifaunal abundance was significantly greater at the West 100 m than the West 500 m site. There was no significant site or season difference for taxa richness of epibenthic macrofauna, and there was no interaction between these factors (Table 3.5, Figure 3.27). The highest taxa richness was at the West and East 100 m sites, in summer, and at the West 100 m and Centre sites, in spring.

Table 3.5 Two-way ANOVA results of motile epibenthic macrofauna data from seven sites along transects, running west and east from a low-density mussel-line aquaculture farm.

Variables	source of variation	df	MS	F ratio	P
Motile epifauna abundance	Site	6	1.110	1.117	0.448
	Season	1	3.214	32.360	0.122
	Site x Season	6	0.993	3.404	0.006
	Residual	56	0.292		
Motile epifauna taxa richness	Site	6	58.833	2.782	0.119
	Season	1	10.414	0.492	0.509
	Site x Season	6	21.148	0.811	0.114
	Residual	56	11.679		

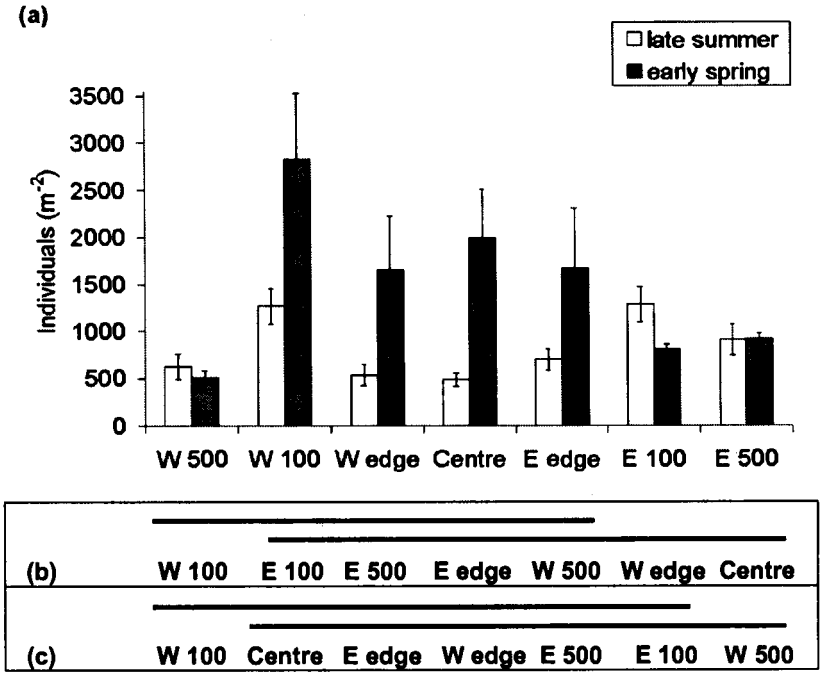


Figure 3.26 (a) Mean motile epifauna abundance (± SE) at 7 sites along transects running west and east from low-density aquaculture at Misery Beach, 2000 (n = 5). **(b)** Tukey's pairwise comparisons of epifauna abundance at sites in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of epifauna abundance at sites in spring ($p < 0.05$).

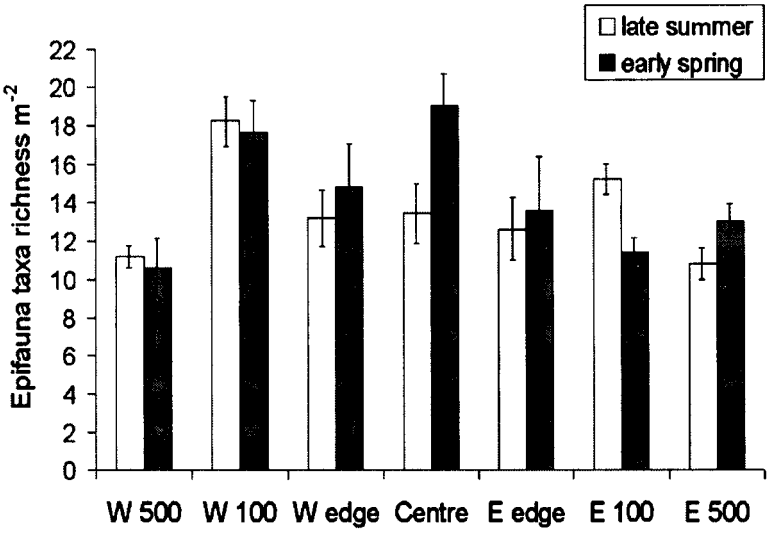


Figure 3.27 Mean motile epifauna taxa richness (± SE) at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 5).

Total taxa richness for motile epifauna was higher at sites adjacent and beneath the farm than more distant sites, suggesting that some species could be associated with the presence of the farm (Table 3.6). No taxa were exclusive to sites away from the farm. Eight taxa were found exclusively at the edge sites, 100 m sites or beneath the farm; three species were only found adjacent to the farm; and one taxon was solely beneath the farm. However, the entire group of Simper selected taxa was found beneath, adjacent and away from the farm.

Table 3.6 Total taxa richness and abundance (m^{-2}) of motile epifauna at sites along transects running west and east from low-density mussel-line aquaculture, in summer. Note: Simper selected taxa and associated abundances are denoted with grey shading (eg. sp. A 10).

Note: Total abundances (m^{-2}) were calculated from the mean number of individuals multiplied by 16, as the sample quadrat size was 0.0625 m^{-2} and were rounded to whole numbers.

Taxa	Away from farm		Adjacent to the farm		Beneath the farm		
	W 500	*E 500*	*W 100*	*E 100*	*W edge*	*E edge*	*Centre*
Anthuridae	13	3	3	3		3	
sp. X	22	58	10	32		6	
Cyproidiedae		3	3	10		6	
Iphimediidae		3		3			
Leucothoidae		6	3	22			13
Aoridae		19	6			10	6
Diogenidae		3	6	3	13		10
Podoceridae	13		6	3	3		3
Gobiesocidae	3	3	6		10		3
Eusiridae	131	131	118	285	22	58	6
Dexaminidae	35	221	253	275	86	189	70
Caprellidae	83	86	45	54	80	48	29
Ischyroceridae	67	90	64	141	42	51	16
L. gammarid sp.	54	173	32	163	22	93	26
Sphaeromatidae	19	13	122	42	42	48	38
Mysidae	70	6	32	26	58	22	22
Hippolytidae	54	26	192	29	29	26	6
Phtisicidae	6	35	3	16	10	19	19
Polychaetae	6	3	58	74	19	35	3
Melitidae	19	3	45	6	10	22	10
Protellidae	22		13	26	19	10	19
Ochlesidae	6		61		6	10	74
Picnogonidae		6	38	6	10	6	13
Nebaliidae		16	54	3	6	3	26
Cumacea sp.			16	3			
Gnathidae			6	3			
sp. S			6				
Phliantidae			13	26	16	3	3
Cypridinidae			3	3	6	6	
Tanaid sp.			13	6	3		16
Stenetrium sp.			10	22		10	3
Majidae			22		13	6	45
Amaryllidae			13		6		3
Arcturidae			10		3		3
Phoxocephalidae			3		3		3
Cylindroleberidae						6	
Total taxa richness	17	21	34	27	25	24	27
Total taxa abundance	627	909	1289	1286	538	698	490

The pattern in motile epifauna composition for spring is similar to those observed for summer (Table 3.7). The majority of individual taxa, and all Simper-selected taxa, were represented beneath the farm, adjacent the farm, and away from the farm, but great variation in the abundances of individual taxon between sites, reveal a gradient in assemblage structure with varying proximity to the farm. Many of the taxa recorded lowest individual abundances away from the farm, highest abundances at the West 100 m site and moderate abundances beneath the farm. Please see Appendix for photos and diagrams of these taxa.

Table 3.7 Total taxa richness and abundance (m^{-2}) of motile epifauna at sites along transects running west and east from a low-density mussel-line aquaculture, in spring. Note: SIMPER selected taxa and associated abundances are denoted with grey shading (eg. sp. A 10). Note: Total abundances (m^{-2}) were calculated from the mean number of individuals multiplied by 16, as the sample quadrat size was 0.0625 $m^{-2,}$ and were rounded to whole numbers.

Taxa	Away from farm		Adjacent the farm		Beneath the farm		
	W 500	*E 500*	*W 100*	*E 100*	*W edge*	*E edge*	*Centre*
Cyproidiedae	80	38	28	32	10	13	10
Caprellidae	99	128	212	236	125	310	6
Mysidae	35	35	12	60	42		22
Iphimediidae	22	6		4	6		10
Arcturidae	10	10	8		6	3	3
Lysianassidae	13		8				
Tanaid sp.			40			10	
Sp. X			68			64	
Gobiesocidae			12	4	10		
Phliantidae		16	28	8	10	19	
Diogenidae		26	16		13	32	
Protellidae		106		124	93		90
Polychaetae	29	48	156		51	54	29
Hippolytidae	35	13	60		35	70	32
Stenetriidae	3	13	88		42	42	32
Anthuridae	6	3	12		10	3	22
Picnogonidae	19	19	56	8	38	29	32
Leucothoidae	26	3		4	19	16	51
Nebaliidae	10	19	196		64	16	93
Dexaminidae	22	96	164	60	186	237	131
Ischyroceridae	42	51	440	32	131	166	246
Melitidae	6	32	536	12	240	154	410
Sphaeromatidae	29	32	456	56	147	51	266
Majidae	6	6	8	4	19	16	16
Podoceridae	6	3	36	8	19	22	42
Eusiridae		195	128	136	176	253	278
Ochlesidae		3	36	8	115	74	29
L. sp.		13	4	16	51	19	16
Isaedae		3	16				67
Cypridinidae	6						22
Gnathidae							3
Cumacean sp.							3
Amaryllidae							32
Total taxa richness	20	24	25	17	24	22	26
Total taxa abundance	506	918	2824	812	1658	1674	1994

3.5 EPIBENTHIC MACROFAUNA: ASSEMBLAGE STRUCTURES AND DIVERSITY: SUMMER

3.5.1 Assemblage patterns of motile epibenthic macrofauna: summer

ANOSIM of the square-root transformed abundance data for motile epifauna showed that there were significant differences in the taxa composition among sites, in summer (P=0.1%, Global R=0.458). Pairwise comparisons indicated that the species composition at the Centre site was significantly different from all other sites, except the West edge (Table 3.8a). Samples from the West and East 500 m sites and the East 100 m site differed significantly to each other and all other sites, while the West and East edge sites were not significantly different from each other, or to the West 100 m site. The nMDS ordination showed a clustering of samples according to their proximity to the mussel farm (Figure 3.28a). Samples from the Centre of the farm formed a broad cluster at the top of the nMDS plot, while samples from the West and East 500 m sites and the East 100 m site, clustered separately from each other at the bottom of the plot. Samples from the East edge site were weakly clustered and also lay towards the bottom right of the plot, and samples from the West edge site and the West 100 m site were weakly clustered between these two broad groups. The clusters formed from the West and East edge samples display the greatest overlap onto samples from the other sites.

When the influence of abundance was removed by using presence/absence data, ANOSIM still showed a significant difference in the species present among sites (P=0.1%, Global R=0.311, Table 3.8b). However, pairwise comparisons showed fewer significant differences than those using square-root transformed data. The West 500 m site differed significantly to all other sites, whereas the East 500 m was not significantly different from either of the 100 m sites, or the West edge. The Centre was significantly different from all other sites. The Centre and the West 500 m sites were the only sites that were significantly different from the West edge site. The nMDS ordination showed a similar pattern to that of the square-root transformed data (Fig. 3.28b). Samples from the West and East 500 m sites clustered separately from each other at the bottom of the nMDS plot. Samples from the Centre of the farm were weakly clustered at the top of the plot. The samples of the West and East 100 m and edge sites were loosely clustered in the middle of the plot. Again, the clusters formed from the West and East edge samples, display the greatest overlap onto samples from the other sites, particularly the centre samples.

Table 3.8 ANOSIM results of data from summer motile epifauna assemblages, from seven sites located along transects running west and east from a low-density mussel-line aquaculture. The ANOSIM results are based on Bray-Curtis dissimilarity matrix, calculated from **(a)** square root and **(b)** presence/absence transformed data. Note: Pairwise comparisons that do not exceed 5 percent are considered significant, and are displayed in bold. All pairwise comparisons are based on 126 permutations.

(a)	W edge		W 100		W 500		E edge		E 100		E 500	
	r value	%	r value	%	r value	%	r value	%	r value	%	r value	%
Centre	0.2	8	0.4	**1**	0.7	**1**	0.7	**1**	0.8	**1**	0.9	**1**
W edge			0.2	7	0.2	**3**	0.2	10	0.5	**1**	0.7	**1**
W 100					0.6	**1**	0.2	7	0.5	**1**	0.7	**1**
W 500							0.4	**1**	0.6	**1**	0.6	**1**
E edge									0.4	**2**	0.4	**2**
E 100											0.4	**1**

(b)	W edge		W 100		W 500		E edge		E 100		E 500	
	r value	%	r value	%	r value	%	r value	%	r value	%	r value	%
Centre	0.1	25	0.2	7	0.6	**1**	0.4	**1**	0.5	**1**	0.6	**1**
W edge			0.0	48	0.3	**2**	0.1	29	0.2	11	0.6	**1**
W 100					0.6	**1**	0.1	20	0.3	6	0.6	**1**
W 500							0.2	9	0.5	**1**	0.6	**1**
E edge									0.1	25	0.3	**3**
E 100											0.5	**1**

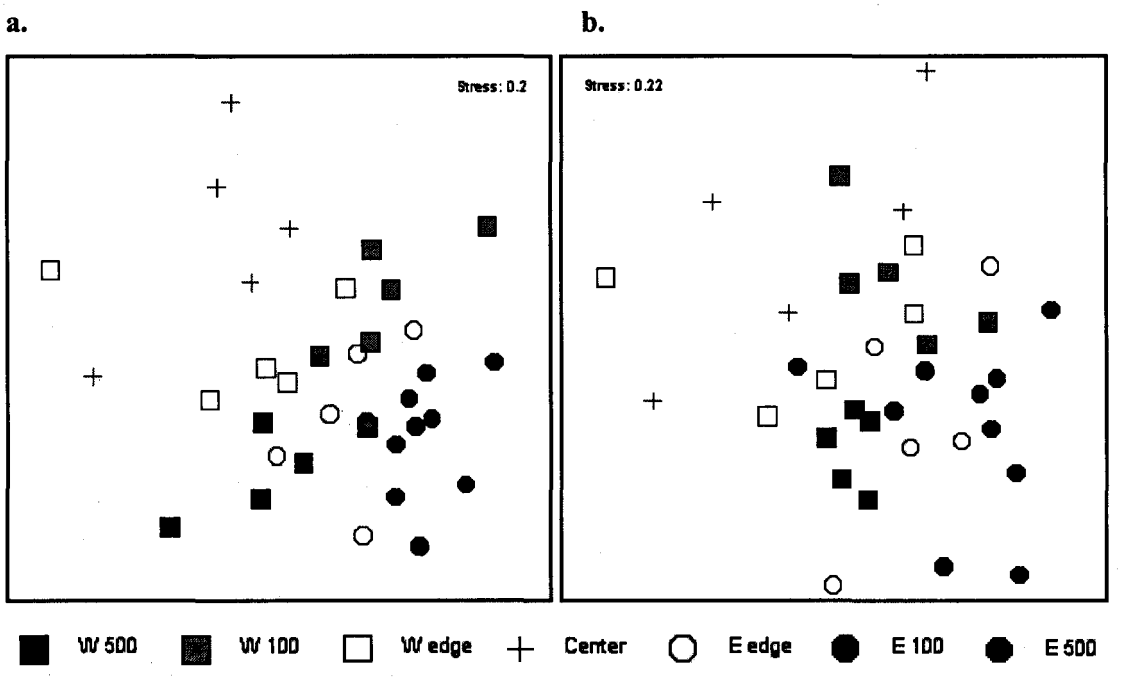


Figure 3.28 Two-dimensional nMDS of summer motile epifauna assemblages from seven sites located along transects running west and east from a low-density mussel-line aquaculture. MDS is based on Bray-Curtis dissimilarity matrix calculated from **(a)** square root transformed data, **(b)** presence/absence data. Note: relatively high stress values, are a reminder that ANOSIM RESULTS must also be considered when interpreting any two-dimensional representation of multi-dimensional data.

Generally, values of Bray-Curtis dissimilarity values between sites, based on summer epibenthic macrofauna assemblages, were smallest between sites close to the mussel farm and greatest between sites close to the farm and those more removed (Table 3.8a, Fig.3.29). Dissimilarity was greatest between the Centre site and the West and East 500 m sites. Values also indicated that the East edge site was more similar to the East 100 m and East 500 m sites than it was to the Centre site, and the West 500 m site was more similar to the West edge site than it was to the Centre site.

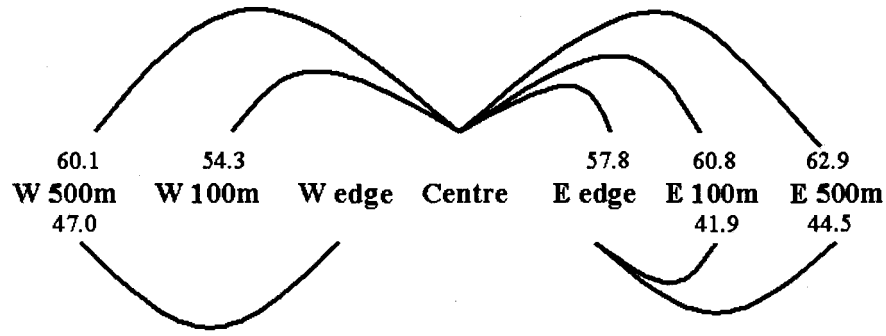


Figure 3.29 Diagrammatic representation of Bray-Curtis dissimilarity values in summer epibenthic macrofauna between sites along transects running west and east from a low-density mussel-line aquaculture, calculated from square root transformed data.

3.5.2 Faunal discriminators of motile epibenthic macrofaunal assemblages: summer

For motile epifauna during summer, Simper analysis showed that low abundance of dexaminid amphipods typically distinguished the West 500 m site from the other sites (Table 3.9). In addition, the abundances of protellid amphipods at the West 500 m site were relatively high compared to the East 500 m site. The East 500 m site was differentiated from the other sites, primarily due to the high abundance of eusirid, ischyrocerid, dexaminid amphipods. Low abundances of eusirid amphipods often distinguished the Centre site from other sites (Table 3.9). However, the relatively high abundance of ochlesid amphipods at the Centre site differentiated it from East 100 m and 500 m sites. The West 100 m site generally differed from other sites due to the relatively high abundances of spheromatid isopods (Table 3.9). In comparison, the East 100 m site was differentiated from the other sites, predominantly by the

high abundances of eusirid, ischyrocerid and dexaminid amphipods, and the absence of ochlesid amphipods. Simper showed the West edge site had a higher abundance; of dexaminid amphipods compared to the West 500 site, and protellid amphipods compared to the East 500 m site. At the East edge site, the abundances of dexaminid and ischyrocerid amphipods were notably high compared with other sites (Table 3.9). Photographs and diagrams of these taxa can be found in the Appendix.

In general, sites away from the farm were differentiated from sites close to the farm by relatively high abundances of eusirid amphipods. Sites close to the farm were differentiated from sites away, by relatively high abundances of dexaminid or protellid amphipods. The Centre site was differentiated from sites away from the farm by the high abundances of ochlesid amphipods (Appendix Figure 3 and Plates 10, 11, 21).

Table 3.9 Simper results of epibenthic macrofauna data for summer, from seven sites along transects, running west and east from a low-density mussel-line aquaculture farm. Simper results were calculated from square root transformed data. Note: Only the taxa with 'average dissimilarity divided by standard deviation' above 2.0 are included.

Sites		Site av. diss.	Family	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
W 500	vs W 100	50	Polychaete taxon 1.	0.3	1.8	3.0	2.4	6.0
	vs W edge	47	Dexaminidae	1.4	2.0	3.3	2.0	7.0
	vs Center	60	Eusiridae	2.7	0.3	6.5	2.1	10.8
	vs E edge	47	Dexaminidae	1.4	3.4	5.0	2.2	10.8
	vs E 100	47	Dexaminidae	1.4	4.1	5.4	2.9	11.6
			Polychaetae	0.3	2.1	3.7	2.3	7.9
	vs E 500	49	Dexaminidae	1.4	3.7	5.5	2.6	11.4
			Protellidae	1.2	0.0	2.9	3.7	6.1
W 100	vs Center	54	Polychaete taxon 1.	1.8	0.2	3.2	2.6	5.8
			Sphaeromatidae	2.3	1.5	2.9	2.0	5.3
	vs E 100	44	Ischyroceridae	1.9	2.6	2.4	2.0	5.4
	vs E 500	52	Polychaetae	1.8	0.2	3.0	2.6	5.7
W edge	vs E 100	50	Eusiridae	0.9	4.1	6.6	2.3	13.1
	vs E 500	57	Taxon 2.	0.0	1.8	4.5	3.1	7.9
			Protellidae	1.1	0.0	2.7	5.7	4.7
Centre	vs E 100	61	Eusiridae	0.3	4.1	7.8	3.4	12.8
			Ischyroceridae	0.8	2.6	4.3	2.3	7.1
			Dexaminidae	2.0	4.1	4.1	2.2	6.7
			Ochlesidae	2.0	0.0	4.0	2.3	6.6
			Polychaete taxon 1.	0.2	2.1	3.9	2.7	6.5
	vs E 500	63	Eusiridae	0.3	2.8	6.4	3.2	10.2
			Ochlesidae	2.0	0.0	4.8	2.2	7.7
			Taxon 2.	0.0	1.8	4.5	3.3	7.1
E edge	vs E 100	42	Ischyroceridae	1.8	2.6	3.1	3.2	7.4
	vs E 500	45	Ischyroceridae	1.8	1.8	3.6	2.4	8.0
E 100	vs E 500	40	Polychaete taxon 1.	2.1	0.2	3.7	2.7	9.1

3.5.3 Environmental determinants of motile epibenthic macrofaunal assemblages: summer

Water column variables were not replicated, and this must be taken into consideration with interpretation of the results. BVSTEP selected porewater ammonium, seagrass leaf maximum height, percentage cover of *Amphibolis antarctica*, and the ratio of ashfree epiphyte to seagrass biomass, as the "best" (most correlated) combination of variables "explaining" the pattern in composition of motile epifauna in summer (Table 3.10). When the composition of epifauna in summer was limited to the most contributing taxa (equating to 95% of the pattern), BVSTEP selected seagrass leaf maximum height and macroalgae % cover as the "best" combination of variables to account for the assemblage pattern of the most contributing epifauna taxa. When the composition of epifauna was limited to taxa identified by Simper analysis, as being responsible for the most significant differences among sites, BVSTEP selected only porewater ammonium (Table 3.12) as the "best" combination of variables to account for the pattern in taxa composition.

In summary, seagrass leaf maximum height (Table 3.10 & 3.11), porewater ammonium (Table 3.10 & 3.12), percentage cover of *Amphibolis antarctica*, the ratio of ashfree epiphyte to seagrass biomass (Table 3.10) and macroalgae %cover (Table 3.11) are shown to be important for "explaining" the patterns in epifaunal composition along the transect at Misery Beach in summer.

Table 3.10 Seagrass ecosystem parameters ranked using BVSTEP according to individual correlations to the taxa composition of motile epifauna, in summer. Note: Variables forming the most significant (P=0.5%) BVSTEP combination, are highlighted.

1 NH_4 in the porewater	**0.545**	17 Nitrite / nitrate (bottom)	0.087
2 Leaf maximum height	**0.538**	18 Chlorophyll-a (surface)	0.052
3 %cover macroalgea	0.418	19 $CaCO_3$ epiphyte / ashfree epiphyte	0.051
4 %cover *Posidonia sinuosa*	0.401	20 Leaf growth (mm per day)	0.022
5 Shoot density	0.351	21 Leaf density	0.008
6 %cover *Amphibolis antarica*	**0.306**	22 Seagrass dry mass	-0.001
7 %cover *Halophila ovalis*	0.284	23 Dissolve inorganic nitrogen (surface)	-0.069
8 $CaCO_3$ epiphyte mass	0.238	24 Total nitrogen (bottom)	-0.101
9 %cover bare sand	0.216	25 Loss on ignition of the sediment	-0.112
10 Ashfree epiphyte / seagrass mass	**0.216**	26 Nitrite / nitrate (surface)	-0.127
11 Depth	0.203	27 Dissolved inorganic nitrogen (bottom)	-0.139
12 Epiphyte dry mass	0.181	28 Chlorophyll-a (bottom)	-0.141
13 $CaCO_3$ epiphyte / seagrass dry mass	0.158	29 NH_4 (bottom)	-0.162
14 Sediment chlorophyll-a	0.125	30 Total nitrogen (surface)	-0.229
15 Ashfree epiphyte mass	0.116	31 NH_4 (surface)	-0.281
16 Leaf area index	0.094	32 Light attenuation	-0.332
BV Step Combination of variables:	**1, 2, 6, 10**	**Rho:**	**0.766**

Table 3.11 Seagrass ecosystem parameters ranked using BVSTEP according to individual correlation to the taxa composition of motile epifauna contributing 95% of the MDS pattern, in summer. Note: Variables forming the most significant (P<0.5%) BVSTEP combination is highlighted.

1	**Leaf maximum height**	**0.636**	17	Sediment chlorophyll 'a'	0.121
2	NH_4 in the porewater	0.549	18	Nitrite / nitrate (bottom)	0.064
3	%cover *Posidonia sinuosa*	0.475	19	%cover *Amphibolis antarica*	0.022
4	Epiphyte dry mass	0.383	20	Seagrass dry mass	-0.004
5	Shoot density	0.368	21	Total nitrogen (surface)	-0.048
6	Depth	0.366	22	Leaf density	-0.066
7	$CaCO_3$ epiphyte mass	0.287	23	Nitrate / nitrate (surface)	-0.081
8	Epiphyte ashfree mass	0.281	24	Leaf growth (mm per day)	-0.082
9	**%cover macroalgea**	**0.280**	25	NH_4 (bottom)	-0.084
10	%cover bare sand	0.235	26	DIN water (bottom)	-0.090
11	$CaCO_3$ epiphyte / seagrass dry mass	0.212	27	Chlorophyll-a (bottom)	-0.095
12	Ashfree epiphyte / seagrass mass	0.210	28	Dissolved inorganic nitrogen (surface)	-0.126
13	$CaCO_3$ epiphyte / ashfree epiphyte	0.196	29	Loss on ignition from sediment	-0.132
14	Chlorophyll-a (surface)	0.171	30	Total nitrogen (bottom)	-0.160
15	Leaf area index	0.169	31	NH_4 (surface)	-0.292
16	%cover *Halophila ovalis*	0.122	32	Light attenuation	-0.364
BV Step Combination of variables:		**1, 9**	**Rho:**		**0.670**

Table 3.12 Seagrass ecosystem parameters ranked according to individual correlation to the taxa composition of the epifauna distinguished by SIMPER analysis, in summer. Note: Variables forming the most significant (P<0.5%) BVSTEP combination is highlighted.

1	**NH_4 (porewater)**	**0.396**	17	%cover *Posidonia sinuosa*	0.021
2	Leaf maximum height	0.301	18	Seagrass dry mass	0.006
3	%cover macroalgea	0.190	19	Leaf density	0.004
4	Depth	0.172	20	NH_4 (bottom)	0.000
5	%cover *Amphibolis antarica*	0.165	21	Chlorophyll 'a' (bottom)	-0.001
6	Sediment chlorophyll 'a'	0.134	22	$CaCO_3$ epiphyte mass	-0.056
7	Nitrite / nitrate (bottom)	0.131	23	Chlorophyll 'a' (surface)	-0.092
8	%cover bare sand	0.110	24	$CaCO_3$ epiphyte / ashfree epiphyte	-0.100
9	Leaf growth (mm per day)	0.093	25	$CaCO_3$ epiphyte / seagrass dry mass	-0.106
10	Ashfree epiphyte / seagrass mass	0.078	26	Total nitrogen (bottom)	-0.129
11	DIN water (bottom)	0.068	27	%cover *Halophila ovalis*	-0.142
12	Epiphyte dry mass	0.065	28	Nitrate / nitrate (surface)	-0.181
13	Dissolved inorganic nitrogen (surface)	0.062	29	Loss on ignition from sediment	-0.207
14	Leaf area index	0.034	30	NH_4 (surface)	-0.258
15	Shoot density	0.031	31	Total nitrogen (surface)	-0.357
16	Epiphyte ashfree mass	0.026	32	Light attenuation	-0.462
BV Step Combination of variables:		**1**	**Rho:**		**0.396**

3.6 BENTHIC MACROFAUNA: ASSEMBLAGE STRUCTURE AND DIVERSITY: SPRING

3.6.1 Assemblage patterns of motile epibenthic macrofauna: spring

During spring, as for the summer data, ANOSIM demonstrated a significant difference in the square-root transformed data for taxa composition of motile epifauna among sites (p=0.1%, Global R=0.422). Pairwise comparisons showed similar results to those for summer data (Table 3.13a). However, in contrast with summer, the spring samples from the West edge site did not differ from the East 500 m and 100 m sites. The nMDS ordination showed a much weaker pattern in spring than that seen for summer (Figure 3.29a vs Figure 3.28a), though samples were still clustered according to their proximity to the mussel farm. The samples from both 500 m sites remained distinct from those at the Centre site. However, there was a greater dissimilarity between samples from the West 500 m and samples from the East 500 m. Samples from the West 500 m site clustered separately in the bottom left quarter of the nMDS plot. Samples from the Centre clustered towards the top of the plot. Most of the other samples lay between those from the West 500 m and Centre sites. In contrast to summer, spring samples from the East 500 m and 100 m sites were clustered, and samples from the West and East edge sites and from the West 100 m site were relatively scattered in the middle of the plot.

Similar to square-root transformed data, ANOSIM reveals a significant difference among sites using presence/absence data (p=0.1%, Global R=0.409). However, pairwise comparisons (Table 3.13b) and nMDS ordination (Figure 3.29b) show a slightly weaker pattern when the influence of abundance is removed by presence/absence transformation of the data. Generally, ordination was clearer and ANOSIM results are more significant using square-root transformed data, indicating that both taxa richness and abundance of motile epifauna were influenced by the proximity to mussel aquaculture.

Data for epifauna show less dissimilarity between sites close to the mussel farm and those more removed (Table 3.13). However, the Centre site was more dissimilar to the East edge site, than it was to the East 100 m or East 500 m site, and the East edge was more dissimilar to the East 500 m than it was to the Centre site (Table 3.13a, Figure 3.30).

Table 3.13 ANOSIM results of data from spring, epifauna assemblages, from seven sites located along transects running west and east from a low-density mussel-line aquaculture farm. The ANOSIM results are based on Bray-curtis dissimilarity matrix, calculated from **(a)** square root and **(b)** presence/absence transformed data. Note: Pairwise comparisons that do not exceed 5% are considered significant, and are displayed in bold. Pairwise comparisons are based on 126 permutations, except for the pairwise comparison, West 100 m vs East 100 m, which is based on 35 permutations.

a	W edge		W 100		W 500		E edge		E 100		E 500	
	r value	%	r value	%	r value	%	r value	%	r value	%	r value	%
Centre	0.2	6	0.5	**1**	0.7	**1**	0.5	**1**	0.9	**1**	0.6	**1**
W edge			0.2	9	0.8	**1**	0.1	29	0.2	9	0.1	29
W 100					0.6	**1**	0.1	15	0.7	**3**	0.5	**2**
W 500							0.6	**1**	0.6	**2**	0.7	**1**
E edge									0.6	**2**	0.4	**1**
E 100											-0.1	83

b	W edge		W 100		W 500		E edge		E 100		E 500	
	r value	%	r value	%	r value	%	r value	%	r value	%	r value	%
Centre	0.6	**1**	0.7	**1**	0.6	**1**	0.5	**1**	0.9	**1**	0.5	**1**
W edge			0.3	6	0.7	**1**	0.2	6	0.3	6	0.1	28
W 100					0.4	**2**	0.1	25	0.7	**3**	0.2	10
W 500							0.5	**1**	0.6	**2**	0.7	**1**
E edge									0.6	**1**	0.4	**1**
E 100											-0.2	83

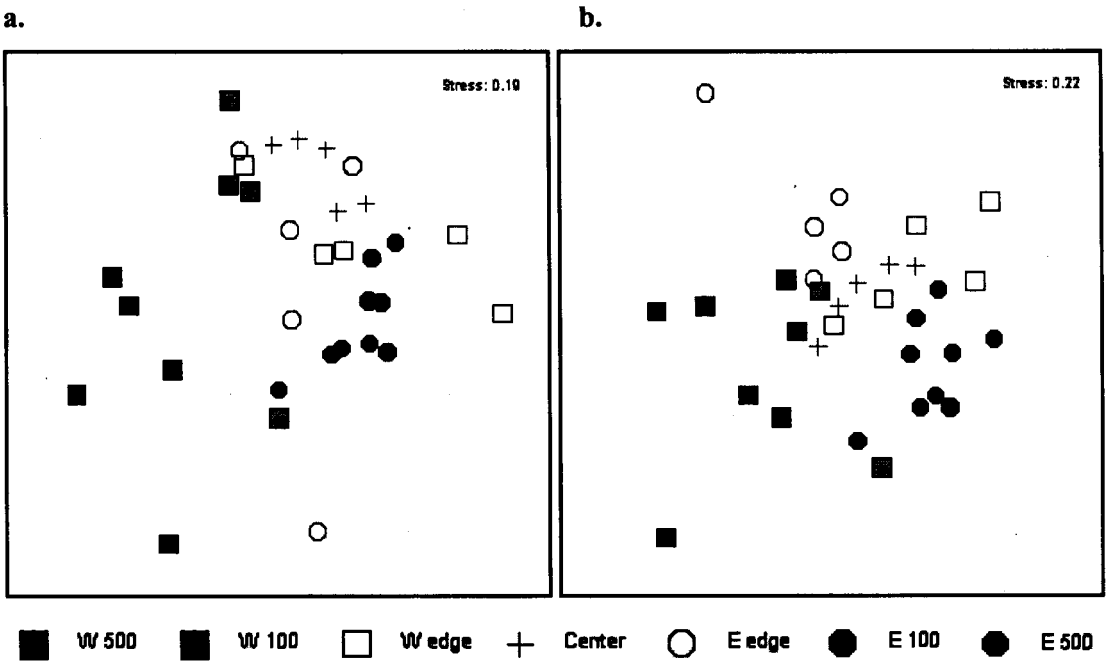


Figure 3.29 Two-dimensional nMDS of spring epibenthic macrofauna assemblages from seven sites located along transects running west and east from low-density mussel-line aquaculture. nMDS is based on Bray-Curtis dissimilarity matrix calculated from **(a)** square root transformed data, **(b)** presence/absence transformed data.

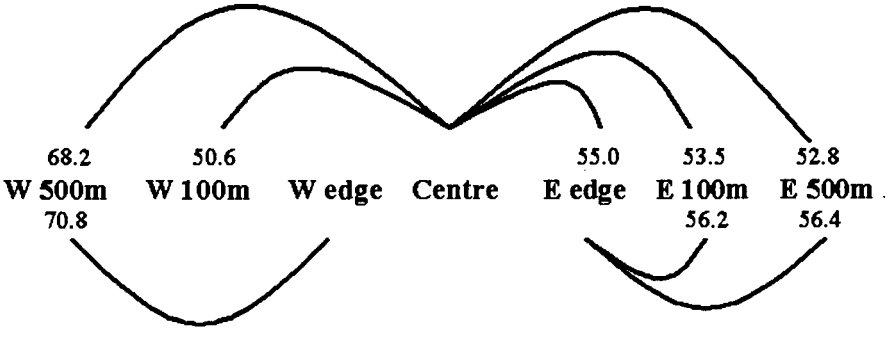


Figure 3.30 Diagrammatic representation of dissimilarity in epifauna assemblages between seven sites along transects running west and east from low-density mussel-line aquaculture, in spring, calculated from square root transformed data.

3.6.2 Faunal discriminators of motile epibenthic macrofaunal assemblages: spring

Simper analysis for spring data shows that for motile epifauna, low abundances of eusirid, ochlesid, dexaminid or protellid amphipods typically distinguished the West 500 m site from all other sites (Table 3.14). Conversely, the East 500 m site is set apart from other sites by relatively high abundances of protellid, eusirid or dexaminid amphipods. The Centre site is distinguished from all sites other than the West edge, due to relatively higher abundances of eusirid, dexaminid, protellid, ochlesid or leucothoid amphipods (Table 3.14). The West 100 m site is often differentiated from other sites by high abundances of Ischyrocerid amphipods, or Stenetriidae, Caprellidae or Sphaeromatidae individuals, while the East 100 m site is distinguished by high abundances of Protellid amphipods or mysid individuals. Simper also shows that a higher abundance of ochlesid and Eusirid amphipods separates the West 500 m site from other sites (Table 3.14). Generally, the East edge site was not different from the West edge and West 100 m sites, but differed from the other sites, by high abundances of eusirid amphipods (Table 3.14). Photographs and diagrams of these taxa can be found in the Appendix.

In summary, sites away from the farm were differentiated from sites close to the farm by relatively low abundances of amphipods. Sites close to the farm were differentiated from sites away, by relatively high abundances of melitid or ischyrocerid amphipods or Hippolytidae, Sphaeromatidae, Caprellidae, or Stenetridae individuals. The Centre site was differentiated from sites away from the farm by the high abundances of eusirid, leucothoid, dexaminid, protellid, or ochlesid amphipods (Appendix Plates 6, 7, 10, 11, 18, 21, 25 and Figures 1, 3, 7).

Table 3.14 Simper results of data from spring epibenthic macrofauna, from seven sites along transects, running west and east from a low-density mussel-line aquaculture farm. The Simper results were calculated from square root transformed data. Note: Only the taxa with 'average dissimilarity divided by standard deviation' above 2.0 are included.

Sites		Site av. diss.	Family	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%
W 500	vs W 100	67	Melitidae	0.4	5.2	7.4	2.2	11.1
			Ischyroceridae	1.4	4.5	5.8	2.5	8.7
			Sphaeromatidae	1.1	4.8	5.5	2.0	8.2
			Eusiridae	0.0	2.7	4.6	3.4	6.8
			Polychaetae	1.2	2.6	3.2	2.0	4.8
			Stenetriidae	0.2	2.2	3.0	2.8	4.5
	vs W edge	71	Eusiridae	0.0	3.2	6.7	3.9	9.5
			Ochlesidae	0.0	2.5	5.0	2.4	7.1
			Taxon 1	0.0	1.7	3.8	2.3	5.4
	vs Center	68	Eusiridae	0.0	3.9	6.6	4.6	9.6
			Dexaminidae	0.5	2.8	4.3	2.1	6.3
			Protellidae	0.0	2.3	4.0	3.5	5.8
			Ochlesidae	0.0	1.3	2.4	3.0	3.6
	vs E edge	68	Eusiridae	0.0	3.9	8.5	4.4	12.5
	vs E 100	63	Eusiridae	0.0	2.9	7.7	5.0	12.3
			Protellidae	0.0	2.6	7.0	2.8	11.2
			Dexaminidae	0.5	1.9	4.6	2.6	7.4
	vs E 500	65	Eusiridae	0.0	3.4	8.4	4.5	12.8
			Protellidae	0.0	2.5	6.0	3.4	9.1
W 100	vs Center	51	Caprellidae	3.4	0.4	3.7	2.1	7.4
			Protellidae	0.0	2.3	2.7	2.5	5.4
			Leucothoidae	0.0	1.7	2.0	2.9	4.0
			Phliantidae	1.3	0.0	1.5	2.6	3.0
	vs E 100	58	Ischyroceridae	4.5	1.4	5.4	3.4	9.2
			Protellidae	0.0	2.6	4.4	2.0	7.5
			Stenetriidae	2.2	0.0	3.1	4.9	5.3
			Mysidae	0.6	1.9	2.0	2.3	3.5
	vs E 500	57	Sphaeromatidae	4.8	1.0	5.2	2.1	9.0
			Ischyroceridae	4.5	1.5	5.1	2.6	8.9
			Protellidae	0.0	2.5	3.8	2.2	6.6
			Stenetriidae	2.2	0.6	2.3	2.0	4.0
Centre	vs E edge	55	Protellidae	2.3	0.0	3.3	2.7	6.0
			Ochlesidae	1.3	1.7	2.1	2.7	3.8
	vs E 100	54	Caprellidae	0.4	3.7	5.6	2.0	10.4
			Leucothoidae	1.7	0.3	2.3	2.4	4.2
			Taxon 1	0.9	0.5	1.6	2.1	3.0
			Mysidae	1.2	1.9	1.2	2.1	2.3
	vs E 500	53	Leucothoidae	1.7	0.2	2.3	2.6	4.3
			Ochlesidae	1.3	0.2	1.8	2.0	3.4
E edge	vs E 100	56	Protellidae	0.0	2.6	5.5	2.1	9.8
			Mysidae	0.0	1.9	4.0	2.8	7.2
			Hippolytidae	2.0	0.0	3.9	3.6	6.9
			Polychaetae	1.6	0.0	3.0	3.2	5.3
			Eusiridae	3.9	2.9	2.3	2.1	4.1
	vs E 500	56	Caprellidae	3.9	2.4	4.7	2.1	8.4
			Protellidae	0.0	2.5	4.7	2.3	8.4
			Polychaetae	1.6	0.8	3.2	2.1	5.6
			Hippolytidae	2.0	0.4	3.1	2.2	5.6
			Mysidae	0.0	1.5	2.9	2.6	5.1

3.6.3 Environmental determinants of motile epibenthic macrofauna assemblage patterns: spring

BVSTEP selected water column surface total nitrogen, sediment loss on ignition, ratio of epiphyte calcium carbonate to ashfree epiphyte biomass, and ammonium in the water column (surface), as the best combination of variables that 'explain' the pattern of taxa composition for epifauna in spring (Table 3.15). When the composition of motile epifauna is limited to the most contributing taxa (equating to 95% of the pattern), total nitrogen in the water column (surface) and percent cover of *P. sinuosa* 'explained' the pattern for epifauna taxa composition using BVSTEP (Table 3.16). When the composition of benthic macrofauna is limited (using SIMPER) to taxa responsible for the most significant difference between sites, BVSTEP selected sediment loss on ignition and percentage cover of *P. sinuosa* (Table 3.17), to account for the patterns in taxa composition.

In summary, water column surface total nitrogen (Table 3.15 &3.16), loss on ignition from the sediment (Table 3.15 & 3.17), percentage cover of *P. sinuosa* (Table 3.16 & 3.17), ammonium in the water column (surface), ratio of calcium carbonate to ash-free epiphyte biomass (Table 3.15) and sediment loss on ignition (Table 3.17) were shown to be important for "explaining" the pattern of taxa composition for epifauna at sites along the transect at the Misery Beach mussel farm in summer.

Table 3.15 Seagrass ecosystem parameters ranked using BVSTEP according to individual correlation to the taxa composition of motile epifauna, in spring. Note: Variables forming the most significant (P<0.5%) BVSTEP combination is highlighted.

1 Total nitrogen (surface)	**0.553**	19 Nitrate/nitrate (bottom)	0.067
2 DIN water (bottom)	0.550	20 Total nitrogen (bottom)	0.056
3 NH_4 (bottom)	0.530	21 Chlorophyll-a sediment	0.048
4 Loss on ignition from the sediment	**0.520**	22 Leaf extention (mm per day)	0.035
5 $CaCO_3$ epiphyte/ashfree epiphyte biom	**0.361**	23 Nitrite/nitrate (porewater)	0.013
6 %cover macroalgea	0.338	24 Light attenuation	0.000
7 %cover *Posidonia sinuosa*	0.338	25 Leaf density	-0.006
8 %cover *Amphibolis antarica*	0.312	26 NH_4 in porewater	-0.029
9 DIN water (surface)	0.281	**27 Chlorophyll-a (bottom)**	**-0.039**
10 NH_4 (surface)	**0.279**	28 DIN porewater	-0.080
11 Epiphyte/seagrass biomass	0.217	29 Ashfree epiphyte biomass	-0.110
12 %cover *Halophila ovalis*	0.210	30 Leaf extention (mg per day)	-0.113
13 Depth	0.203	31 $CaCO_3$ epiphyte biomass	-0.145
14 %cover bare sand	0.200	32 Seagrass biomass	-0.192
15 Leaf maximum height	0.172	33 Shoot density	-0.244
16 Epiphyte biomass	0.160	34 Leaf Area Index	-0.190
17 Ashfree epiphyte/seagrass biomass	0.095	35 Nitrite/nitrate (surface)	-0.301
18 Chlorophyll-a (surface)	0.078	36 $CaCO_3$ epiphyte/ashfree epiphyte biom	-0.315
BV Step Combination of variables: 1, 4, 5, 10, 27		**Rho: 0.808**	

Table 3.16 Seagrass ecosystem parameters ranked according to individual correlation to the taxa composition of the motile epibenthic macrofauna contributing 95% of the MDS pattern, in spring.
Note: Variables forming the most significant (P<0.5%) BVSTEP combination is highlighted.

1 Total nitrogen (surface)	**0.582**	19 Leaf maximum height	0.060
2 DIN water (bottom)	0.478	20 Chlorophyll-a (surface)	0.028
3 NH_4 (bottom)	0.459	21 Light attenuation	0.022
4 Loss on ignition from sediment	0.398	22 Nitrite/nitrate (bottom)	0.000
5 Epiphyte/seagrass biomass	0.296	23 Nitrite/nitrate (porewater)	-0.005
6 Depth	0.275	24 Total nitrogen (bottom)	-0.013
7 %cover *Posidonia sinuosa*	**0.266**	25 Chlorophyll-a (bottom)	-0.015
8 Epiphyte biomass	0.243	26 $CaCO_3$ epiphyte mass	-0.025
9 $CaCO_3$ epiphyte/ashfree epiphyte	0.227	27 Leaf density	-0.032
10 %cover macroalgea	0.215	28 Ashfree epiphyte biomass	-0.057
11 %cover *Amphibolis antarica*	0.194	29 NH4 in porewater	-0.070
12 NH_4 (surface)	0.135	30 DIN porewater	-0.102
13 Dissolved inorganic nitrogen (surface)	0.127	31 Seagrass biomass	-0.113
14 %cover bare sand	0.122	32 Leaf extention (mg per day)	-0.145
15 %cover *Halophila ovalis*	0.094	33 Leaf Area Index	-0.216
16 Ashfree epiphyte/seagrass mass	0.074	34 $CaCO_3$ epiphyte/seagrass biomass	-0.218
17 Leaf extention (mm per day)	0.070	35 Shoot density	-0.226
18 Chlorophyll-a in the sediment	0.068	36 Nitrate/nitrate (surface)	-0.251
BV Step Combination of variables:	**1, 7**	**Rho:**	**0.670**

Table 3.17 Seagrass ecosystem parameters ranked according to individual correlation to the composition of the motile epifauna taxa distinguished by Simper analysis, in spring.
Note: Variables forming the most significant (P<0.5%) BVSTEP combination is highlighted.

1 Loss on ignition from the sediment	**0.570**	19 Epiphyte biomass	0.081
2 %cover *Posidonia sinuosa*	**0.521**	20 Epiphyte/seagrass biomass	0.073
3 %cover macroalgea	0.511	21 Chlorophyll-a sediment	0.035
4 DIN water (bottom)	0.474	22 Nitrite/nitrate (bottom)	0.033
5 Total nitrogen (surface)	0.447	23 Ashfree epiphyte/seagrass biomass	0.019
6 NH_4 (bottom)	0.442	24 Nitrite / nitrate (porewater)	-0.049
7 %cover *Amphibolis antarica*	0.437	25 Leaf extention (mm per day)	-0.056
8 %cover bare sand	0.387	26 Leaf density	-0.078
9 Leaf maximum height	0.379	27 Nitrate/nitrate (surface)	-0.120
10	0.373	28 Chlorophyll-a (bottom)	-0.135
11 Chlorophyll-a (surface)	0.303	29 Light attenuation	-0.146
12 %cover *Halophila ovalis*	0.297	30 Ashfree epiphyte biomass	-0.158
13 Dissolved inorganic nitrogen (surface)	0.262	31 Leaf extention (mg per day)	-0.173
14 NH_4 (surface)	**0.241**	32 $CaCO_3$ epiphyte biomass	-0.200
15 NH_4 in porewater	0.184	33 Shoot density	-0.209
16 Total nitrogen (bottom)	0.152	34 Leaf Area Index	-0.230
17 DIN porewater	0.117	35 Seagrass dry mass	-0.266
18 Depth	0.099	36 $CaCO_3$ epiphyte/seagrass dry mass	-0.277
BV Step Combination of variables:	**1, 2, 14**	**Rho:**	**0.752**

3.7 BENTHIC MACROINFAUNA: TAXA RICHNESS AND DENSITIES

There was a significant season effect and site x season interaction for infauna abundance and taxa richness (Table 3.18). In summer, abundance and taxa richness of infauna at the West edge and Centre sites were shown by Tukey's test to be significantly lower than at other sites, reflecting the zero catches at these two sites (Figures 3.31 & 3.32). Additionally, there were significantly greater abundances and taxa richness at the West 100 m site than at the East edge site. Compared to summer, the trend for infauna in spring was comparable; however the peak had relocated from the West 100m site to the East 100 m site, which was greater than at the Centre site. Taxa richness of infauna during spring was significantly greater at the East 500 m site than at the Centre site.

Table 3.18 Two-way ANOVA results of benthic macroinfauna data from seven sites along transects, running west and east from a low-density mussel-line aquaculture farm.

Variables	source of variation	df	MS	F ratio	P
Infauna abundance	Site	6	32.970	1.251	0.396
	Season	1	0.135	0.985	0.359
	Site x Season	6	26.362	91.208	0.001
	Residual	56	0.289		
Infauna taxa richness	Site	6	5.115	1.226	0.406
	Season	1	0.095	0.023	0.885
	Site x Season	6	4.173	18.631	0.001
	Residual	56	0.224		

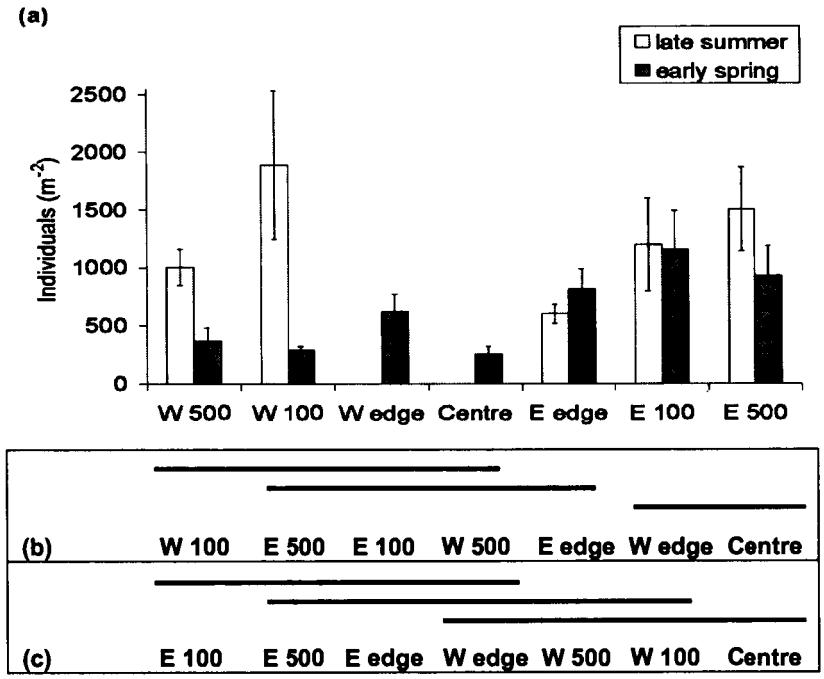


Figure 3.31 **(a)** Mean infauna abundance (± SE) at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n=5). **(b)** Tukey's pairwise comparisons of infauna abundance at sites in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of infauna abundance at sites in spring ($p < 0.05$).

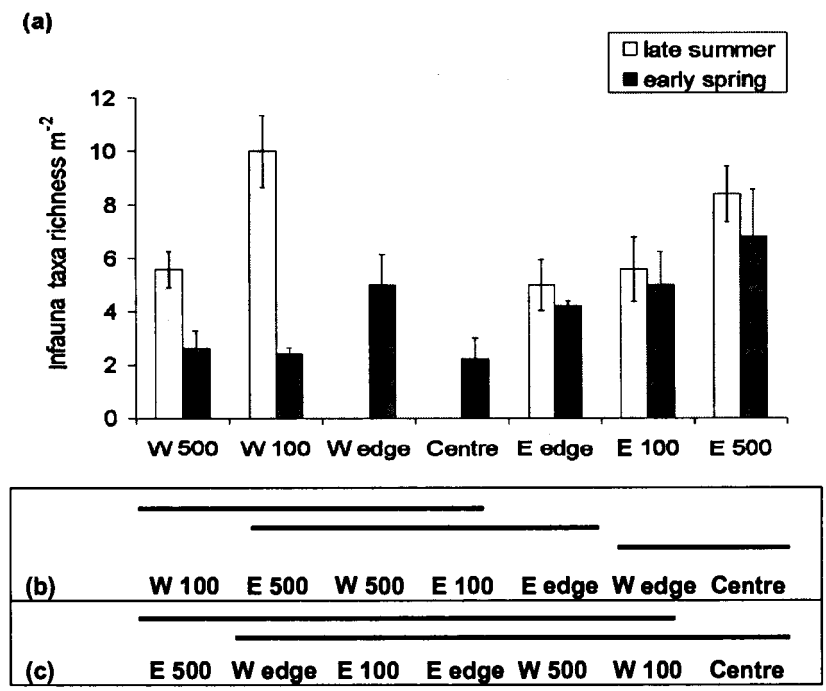


Figure 3.32 **(a)** Mean infauna taxa richness (± SE) at 7 sites along transects running west and east from low-density aquaculture at Misery Beach (n = 5). **(b)** Tukey's pairwise comparisons of infauna richness at sites in summer; sites are reordered, highest to lowest from left to right and lines denote sites not significantly different from each other ($p < 0.05$). **(c)** Tukey's pairwise comparisons of infauna richness at sites in spring ($p < 0.05$).

During summer, no infauna was recorded at the West edge and Centre sites, while infauna abundances at the East edge site was relatively low (Table 3.19). Taxa richness was highest at the West 100 m site. Out of 31 taxa, 13 taxa representing the majority of individuals were found beneath, adjacent to and away from the farm. Of these 13 taxa, eight were found adjacent to and away from the farm, but not beneath, while five taxa were found exclusively adjacent the farm and Amphinomidae, Serpulidae and Nephtyidae individuals were found exclusively beneath the farm. One taxon was found solely beneath and adjacent to the farm, while Sphaerodoridae and Nemertean individuals were found exclusively at sites away from the farm (Table 3.19). Five of the six Simper selected taxa were found in all three zones of proximity to the farm.

Table 3.19 Total taxa richness and abundance ($m^{-2)}$ of benthic macro-infauna at sites along transects running west and east from low-density mussel-line aquaculture, in summer. Note: Simper selected taxa and associated abundances are denoted with grey shading (eg. sp. A 10). Note: Total abundances (m^{-2}) were calculated from the mean number of individuals multiplied by 16, as the sample quadrat size was 0.0625 m^{-2}, and were rounded to whole numbers.

Taxa	Away from farm		Adjacent the farm		Beneath the farm		
	W 500	*E 500*	*W 100*	*E 100*	*E edge*	*W edge*	*Centre*
Sphaerodoridae	24						
Nemertea sp.	24	39					
Anthuridae	72	19		19			
Chrysopetalidae	72		77				
Phoxocephalidae		96	154				
Maldanidae		19	19				
Trichobranchidae		19	39				
Opheliidae		39	19		24		
Cirratulidae		39	77		24		
Oweniidae	48		19		24		
Nereididae	24	39		39	24		
Eunicidae	193	39	19	58			
Ischerochidae	72	39	58	19			
Sipunicula sp.	24	19	39	39			
Cypridinodes sp.	48	19	19	19	24		
Majidae	144	58	96	58	48		
Polynoidae	120	539	250	539	24		
Syllidae	120	77	173	58	72		
Sabellidae	24	39	19	96	72		
Capitellidae		116	173	58	48		
Paronidae		193	154	39	48		
Spionidae		58	250	19	48		
Terebellidae			39	19			
Cumacea sp.			77				
Gnathiidae			39				
Lumbrinereidae			96				
Euphrosinidae				19			
Ampeliscidae				77	24		
Amphinomidae					24		
Serpulidae					48		
Nephtyidae					24		
Total taxa richness	14	19	22	16	15	0	0
Total taxa abundan	1011	1502	1906	1175	602	0	0

Similar to summer, most of the infauna taxa (and all Simper selected taxa) found beneath and adjacent to the farm in spring, were also found away from the farm (Table 20). However, Nereididae, Terebellidae, Nephtyidae and Echiua individuals, which were found away from the farm, were not found adjacent or beneath the farm. Conversely, Nereididae, Amphithoidae, Cypronoides, Gnathiidae, Anthuridae individuals found beneath or adjacent to the farm were not found at sites distant from the farm. Taxa richness was highest away from the farm (East 500 m site) and lowest beneath the farm (Centre site), Differences in macroinfauna taxa composition, between sites of varying proximity are apparent.

Table 3.20 Total taxa richness and abundance (m^{-2}) of infauna at sites along transects running west and east from a low-density mussel-line aquaculture the mussel farm, in spring. Note: Simper selected taxa and associated abundances are denoted with grey shading (eg. sp. A 10). Note: Total abundances (m^{-2}) were calculated from the mean number of individuals multiplied by 16, as the sample quadrat size was 0.0625 m^{-2}, and were rounded to whole numbers.

Taxa	Away from farm		Adjacent the farm		Beneath the farm		
	W 500	*E 500*	*W 100*	*E 100*	*W edge*	*E edge*	*Centre*
Nereididae	39						
Terebellidae		24					
Echiua sp.		24					
Nephtyidae		96					
Nemertea sp.	19	48	24				
Cirratulidae		24	24				
Polynoidae		24	24	24			
Sipunculan		24		96			
Majidae	39	24		24	72		
Eunicidae	58	72		72	48	48	
Serpulidae		72		361		217	
Ophelidae		24			24	24	
Phoxocephalidae		48		48	48	144	
Capitellidae		72		72	24	24	
Nereididae			24	72	48		
Ampithoidae				48	48	96	
Cypridinodes sp.			24			72	19
Paronidae		24		24		72	19
Chrysopetalidae	19	168	24		24	24	19
Ischerochidae	19	72	24	120	96		116
Syllidae	135	48	48	48	72	72	19
Lumbrineridae	39		72	48	96	24	39
Gnathiidae					24		
Anthuridae							19
Total taxa richness	8	17	9	13	12	11	7
Total taxa abundance	366	890	289	1059	626	818	250

3.8 BENTHIC MACROINFAUNA: ASSEMBLAGE STRUCTURE AND DIVERSITY: SUMMER

3.8.1 Assemblage patterns of benthic macroinfauna: summer

It is important to note that for summer infauna, the Centre and West edge sites were removed from the nMDS ordination and the ANOSIM analyses because no infauna were found in the samples from these two sites, and PRIMER does not allow inclusion of the sites, recording zero values only, for the analysis. However, the absence of infauna in these samples suggests that the Centre and West edge sites differ from the other sites where infauna was present. ANOSIM of the square-root transformed abundance data for infauna showed that there was a significant difference in the taxa composition among sites (P=0.1%, Global R=0.288). Pairwise comparisons indicated that samples of the West 500 m differed significantly to the West 100 m, East 100 m and East 500 m sites (Table 3.21a). The East edge and East 500 m sites were not significantly different from other sites. The West 100 m site was significantly different from the East 100 m site. nMDS ordination showed some degree of clustering of samples from the 500 m and 100 m sites and the East edge site (Fig. 3.34a). Samples from the West 500 m site formed the tightest cluster towards the left of the nMDS plot, while samples from the West 100 m clustered towards the middle of the plot. Samples from the East 500 m site were also clustered in the middle of the plot, while samples from the East 100 m clustered at the bottom, and samples from the East edge were intermingled.

When the influence of abundance is removed by using presence/absence data, ANOSIM still shows a significant difference among sites (P=0.1%, Global R=0.244). However, pairwise comparisons showed some slight alterations to the significance of some differences (Table 3.21b). In comparison to square-root transformed data, presence/absence data showed that the West 500 m site was not significantly different from the East 100 m site. In addition, the West 100 m site was significantly different from the East edge site, and the West edge and Centre sites, which both recorded zero values for abundance. The nMDS ordination of presence/absence data shows a similar pattern to that for the square root transformed data (Fig. 3.34b). Samples from the East 500 m site clustered weakly near the middle of the nMDS plot, while samples from the West 500 m site were clustered on the left of the plot. Samples from the East 100 m site clustered at the bottom, while samples from the West 100 m site clustered towards the middle of the plot, and samples from the East edge site were scattered around the plot.

In general, the data from summer infauna assemblages show lowest dissimilarity values between sites close to the mussel farm, and greater dissimilarity between sites that were more distant from each other. It is important to consider the absence of infauna at some of the sites when interpreting the ANOSIM results. Since no infauna was present in the samples from the Centre and West edge sites, they were omitted from the analysis. However, it may be interpreted that the West 100 m and East 500 m sites, which had highest taxa richness and abundance, were the most dissimilar to the Centre and West edge site, which recorded zero for taxa richness and abundance. For those sites where the fauna were sampled, the dissimilarity values were greatest between West 500 m and East 500 m sites (Fig. 3.34, Table 3.21). The West 500 m and East edge sites were the least dissimilar for infauna composition.

Table 3.21 ANOSIM results of data from summer infauna assemblages, from seven sites along transects running west and east from a low-density mussel-line aquaculture farm. The ANOSIM results are based on Bray-Curtis dissimilarity matrix, calculated from **(a)** square root and **(b)** presence/absence transformed data. Note: Pairwise comparisons that do not exceed 5 percent are considered significant, and are displayed in bold. Five asterisks (*****) represents the absence of a result for particular sites where zero values where inappropriate for ANOSIM. All pairwise comparisons are based on 126 permutations, except for the West 500 m comparison with East edge, which is based on 35 permutations.

a	W edge		W 100		W 500		E edge		E 100		E 500	
	r value	%	r value	%	r value	%	r value	%	r value	%	r value	%
Centre	****	****	****	****	****	****	****	****	****	****	****	****
W edge			****	****	****	****	****	****	****	****	****	****
W 100					0.7	1	0.3	6	0.4	1	-0.2	90
W 500							0.4	9	0.5	**2**	0.6	**1**
E edge									0.2	6	0.1	28
E 100											0.2	9

b	W edge		W 100		W 500		E edge		E 100		E 500	
	r value	%	r value	%	r value	%	r value	%	r value	%	r value	%
Centre	****	****	****	****	****	****	****	****	****	****	****	****
W edge			****	****	****	****	****	****	****	****	****	****
W 100					0.7	1	0.3	**5**	0.4	**1**	-0.1	71
W 500							0.3	17	0.4	**3**	0.5	**2**
E edge									0.1	29	0.1	39
E 100											0.1	23

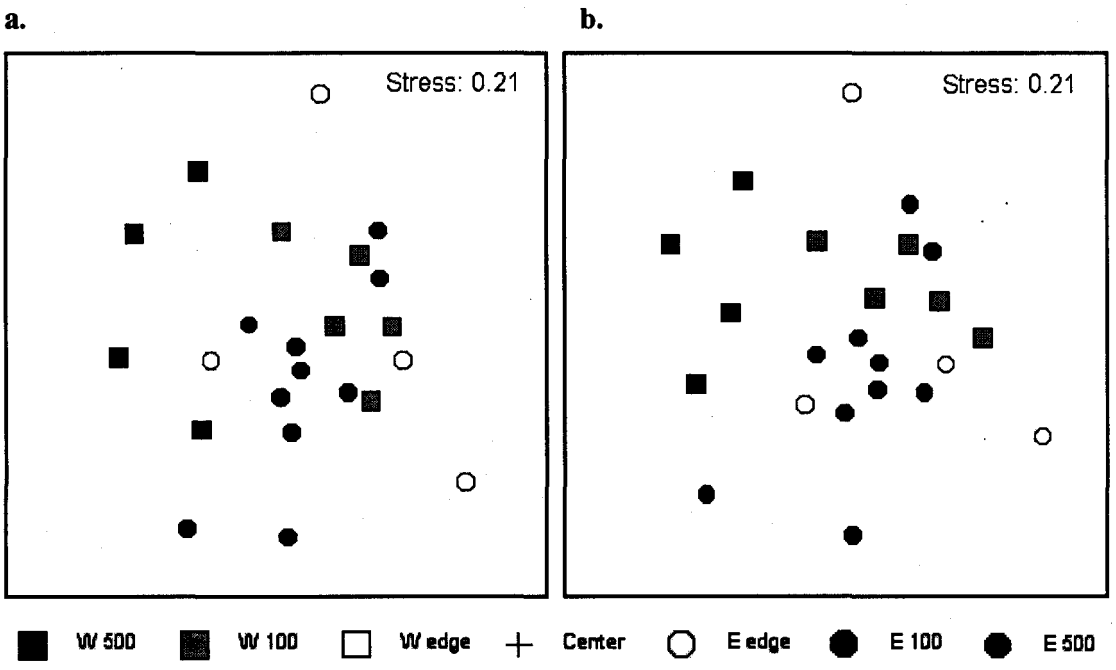


Figure 3.34 Two-dimentional nMDS of summer infauna assemblages from seven sites along transects running west and east from a low-density mussel-line aquaculture. nMDS is based on Bray-Curtis dissimilarity matrix calculated from **(a)** square root transformed data, **(b)** presence/absence transformed data.

3.8.2 Faunal discriminators of benthic macroinfaunal assemblages: summer

For infauna during summer, it is again important to note that zero values were recorded for infauna at the Centre site and West edge site. Thus, these two sites were distinct from the other five sites, particularly the East and West, 100 m and 500 m sites where substantial abundances of infauna were recorded.

Simper analysis for those sites where fauna was present show that, in general, high abundance of syllid and eunicid polychaetes differentiated the West 500 m site from other sites (Table 3.22). In contrast, the West 100 m site had a high abundance of capitellid, paronid, spoinid and syllid polychaetes. The East 500 m site was characterised by high abundances of capitellid and paronid polychaetes, while the East 100 m site had predominantly high abundances of polynoid polychaetes, which in most cases differentiated it from other sites. Simper highlights the generally low abundances of infauna at the East edge site (Table 3.22).

In general, sites away from the farm were differentiated from sites close to the farm by relatively high abundances of eunicid polychaetes. Sites close to the farm were differentiated from sites away, by relatively high abundances of capitellid, polynoid, spoinid or syllid polychaetes.

Table 3.22 Simper results of infauna data from summer, at seven sites along transect, running west and east from a low-density mussel-line aquaculture. The Simper results were calculated from square root transformed data. Note: Only the species with 'average dissimilarity divided by standard deviation' above 1.4 are included. The Centre site and West edge site were omitted completely as no infauna was present in the samples from these sites.

Sites		Site Av. Diss.	Family	Av. Abund.	Av. Abund.	Av. Diss.	Diss. / SD	Contrib. %
W 500	vs W 100	82	Syllidae	1.3	1.8	5.9	1.7	7
			Capitellidae	0.0	1.8	5.8	1.7	7
			Spoinidae	0.0	2.6	5.8	1.8	7
			Eunicidae	2.0	0.2	5.7	1.5	7
			Paronidae	0.0	1.6	5.4	1.8	7
	vs E 100	80	Polynoidae	1.3	5.6	12.7	2.3	16
	vs E 500	80	Paronidae	0.0	2.0	7.7	1.7	10
			Capitellidae	0.0	1.2	5.8	1.9	7
			Eunicidae	2.0	0.4	5.8	1.5	7
W 100	vs E edge	75	Spoinidae	2.6	0.5	5.6	1.6	8
			Paronidae	1.6	0.5	5.2	1.4	7
	vs E 100	73	Polynoidae	2.6	5.6	7.0	1.5	10
			Spoinidae	2.6	0.2	5.3	1.5	7
			Paronidae	1.6	0.4	4.9	1.4	7
			Sabellidae	0.2	1.0	3.9	1.4	5

3.8.3 Environmental determinants of benthic macroinfaunal assemblage patterns: summer

For infauna in summer, BVSTEP selected the ratio of epiphyte calcium carbonate content to seagrass biomass, water column-surface chlorophyll-a, and water-column surface total nitrogen, as the best combination of variables "explaining" the pattern of infauna composition of infauna in summer (Table 3.23). When the composition of infauna was limited to the most contributing taxa (equating to 95% of the pattern), BVSTEP selected the ratio of epiphyte calcium carbonate to seagrass biomass, and water-column surface nitrate/nitrite, as the "best" combination of variables to account for the assemblage pattern of the most contributing infauna taxa (Table 3.24). When the composition of infauna was limited to taxa selected by SIMPER analysis as being responsible for the most significant differences among sites, BVSTEP again selected the ratio of epiphyte calcium carbonate to seagrass biomass, and water-column surface nitrate/nitrite, as the "best" combination of variables to account for the pattern in taxa composition (Table 3.25).

In summary, the ratio of epiphyte calcium carbonate content to seagrass biomass (Tables 3.23, 3.24 & 3.25), water-column surface nitrate-nitrite (Tables 3.24 & 3.25), water-column surface total nitrogen, and water column surface chlorophyll-a (Table 3.23) are shown to be important variables for "explaining" patterns for taxa composition of infauna, in summer.

Table 3.23 Seagrass ecosystem parameters ranked using BVSTEP according to individual correlations to the taxa composition of infauna, in summer. Note: Variables forming the most significant (P<0.5%) BVSTEP combination is highlighted.

1 $CaCO_3$ epiphyte/seagrass biomass	**0.697**	17 Dissolved inorganic nitrogen (surface)	-0.152
2 %cover *Posidonia sinuosa*	0.685	18 Depth	-0.164
3 Nitrate/nitrate (surface)	0.648	19 Nitrite/nitrate (bottom)	-0.188
4 %cover *Halophila ovalis*	0.606	20 NH_4 (surface)	-0.274
5 Chlorophyll-a (surface)	**0.575**	21 Total nitrogen (bottom)	-0.285
6 %cover macroalgea	0.535	22 NH_4 (bottom)	-0.309
7 %cover *Amphibolis antarica*	0.479	23 Shoot density	-0.321
8 $CaCO_3$ epiphyte/ashfree epiphyte	0.395	24 $CaCO_3$ epiphyte biomass	-0.418
9 Ashfree epiphyte/seagrass biomass	0.333	25 DIN water (bottom)	-0.418
10 Light attenuation	0.330	26 Chlorophyll-a (bottom)	-0.439
11 Total nitrogen (surface)	**0.297**	27 Epiphyte ashfree biomass	-0.491
12 NH_4 (porewater)	0.164	28 Epiphyte biomass	-0.648
13 %cover bare sand	0.127	29 Leaf growth (mm per day)	-0.703
14 Sediment chlorophyll-a	0.115	30 Leaf area index	-0.818
15 Loss on ignition from the sediment	-0.006	31 Leaf density	-0.830
16 Leaf maximum height	-0.134	32 Seagrass biomass	-0.891
BV Step Combination of variables:	**1, 5, 11**	**Rho:**	**0.976**

Table 3.24 Seagrass ecosystem parameters ranked using BVSTEP according to individual correlations to the taxa composition of the infauna contributing 95% of the pattern in summer. Note: Variables forming the most significant (P<0.5%) BVSTEP combination is highlighted.

1 $CaCO_3$ epiphyte/seagrass biomass	**0.782**	17 Depth	-0.079
2 %cover *Posidonia sinuosa*	0.697	18 Dissolved inorganic nitrogen (surface)	-0.139
3 %cover macroalgea	0.673	19 Nitrite/nitrate (bottom)	-0.139
4 %cover *Halophila ovalis*	0.661	20 NH_4 (surface)	-0.201
5 Nitrate/nitrate (surface)	**0.588**	21 Total nitrogen (bottom)	-0.212
6 %cover *Amphibolis antarica*	0.552	22 NH_4 (bottom)	-0.273
7 Chlorophyll-a (surface)	0.422	23 Shoot density	-0.273
8 Ashfree epiphyte/seagrass mass	0.418	24 DIN water (bottom)	-0.345
9 Light attenuation	0.251	25 Chlorophyll-a (bottom)	-0.390
10 NH_4 (porewater)	0.224	26 $CaCO_3$ epiphyte mass	-0.418
11 Total nitrogen (surface)	0.212	27 Epiphyte ashfree biomass	-0.564
12 %cover bare sand	0.164	28 Leaf growth (mm per day)	-0.630
13 Chlorophyll-a in the sediment	0.127	29 Epiphyte biomass	-0.648
14 Loss on ignition from the sediment	0.018	30 Leaf area index	-0.745
15 $CaCO_3$ epiphyte/ashfree epiphyte	-0.042	31 Leaf density	-0.794
16 Leaf maximum height	-0.061	32 Seagrass biomass	-0.855
BV Step Combination of variables:	**1, 5**	**Rho:**	**0.915**

Table 3.25 Seagrass ecosystem parameters ranked according to individual correlations to the composition of the most important infauna taxa determined by Simper analysis, in summer. Note: Variables forming the most significant (P<0.5%) BVSTEP combination is highlighted.

1 CaCO3 epiphyte/seagrass biomass	**0.782**	17 Depth	-0.164
2 %cover *Halophila ovalis*	0.758	18 Shoot density	-0.188
3 %cover macroalgea	0.699	19 $CaCO_3$ epiphyte/ashfree epiphyte	-0.212
4 %cover *Posidonia sinuosa*	0.685	20 Dissolved inorganic nitrogen (surface)	-0.236
5 Nitrate/nitrate (surface)	**0.552**	21 Total nitrogen (bottom)	-0.248
6 %cover *Amphibolis antarica*	0.527	22 Chlorophyll-a (bottom)	-0.250
7 Chlorophyll-a (surface)	0.465	23 NH_4 (surface)	-0.267
8 Ashfree epiphyte/seagrass mass	0.405	24 NH_4 (bottom)	-0.273
9 NH_4 (porewater)	0.370	25 DIN water (bottom)	-0.297
10 Light attenuation	0.312	26 $CaCO_3$ epiphyte mass	-0.297
11 Total nitrogen (surface)	0.127	27 Epiphyte ashfree biomass	-0.503
12 %cover bare sand	0.067	28 Leaf growth (mm per day)	-0.538
13 Loss on ignition from the sediment	-0.006	29 Epiphyte biomass	-0.576
14 Chlorophyll-a in the sediment	-0.042	30 Leaf area index	-0.661
15 Nitrite/nitrate (bottom)	-0.115	31 Leaf density	-0.733
16 Leaf maximum height	-0.116	32 Seagrass biomass	-0.758
BV Step Combination of variables:	**1, 5**	**Rho:**	**0.903**

3.9 BENTHIC MACROINFAUNA: ASSEMBLAGE STRUCTURE AND DIVERSITY: SPRING

3.9.1 Assemblage patterns of benthic macroinfauna: spring

For infauna collected in spring, ANOSIM of the square-root transformed abundance data shows a significant difference in taxa composition among sites (p=1.1%, Global R=0.186). Pairwise comparisons show that the Centre site was significantly different from the West and East edge sites and East 100 m sites (Table 3.26a). The nMDS ordination show weak clustering of samples according to their proximity to the mussel farm (Fig. 3.35a). Samples from the Centre and West edge sites are clustered weakly towards the bottom of the nMDS plot, while those from the East edge site are clustered towards the top of the nMDS plot. All other samples are intermingled around these sites. For presence/absence data, ANOSIM also showed that there was a significant difference among sites (P=0.8%, Global R=0.191). Pairwise comparisons showed a significant difference between the West 100 m and East edge sites, and between the West 500 m and East 100 m sites, but no significant difference between the Centre and West edge sites (Table 3.26). The MDS ordination showed a similar pattern to that for square-root transformed data (Fig. 3.35b).

Table 3.26 ANOSIM results of data from spring infauna assemblages, from seven sites along transects running west and east from a low-density mussel-line aquaculture farm. The ANOSIM results were based on Bray-curtis dissimilarity matrix, calculated from **(a)** square root and **(b)** presence/absence transformed data. Note: Pairwise comparisons that did not exceed 5% were considered significant, and are displayed in bold. Five asterisks (*****) represents the absence of a results for particular sites where zero values where inappropriate for ANOSIM. Pairwise comparisons were based on 126 permutations, except the West 100 m comparison with West edge, West 100 m vs East edge, West 100 m vs East 100 m, West edge vs East edge, West edge vs East 100 m, and East edge vs East 100 m, which were based on 35 permutations.

a

	W edge		**W 100**		**W 500**		**E edge**		**E 100**		**E 500**	
	r value	%	r value	%	r value	%	r value	%	r value	%	r value	%
Centre	0.1	**2**	0.0	36	0.1	20	0.4	**3**	0.4	**2**	0.2	10
W edge			-0.1	71	-0.1	68	0.2	6	0.1	29	0.1	31
W 100					-0.2	87	0.0	49	0.3	14	-0.1	71
W 500							0.0	46	0.3	6	0.0	57
E edge									0.3	17	0.0	60
E 100											0.1	26

b

	W edge		**W 100**		**W 500**		**E edge**		**E 100**		**E 500**	
	r value	%	r value	%	r value	%	r value	%	r value	%	r value	%
Centre	0.1	29	0.0	35	0.1	19	0.4	**3**	0.5	**1**	0.2	10
W edge			-0.1	65	-0.1	64	0.3	17	0.2	17	0.1	26
W 100					-0.2	83	0.0	**5**	0.3	9	-0.1	71
W 500							0.0	44	0.3	**5**	0.0	56
E edge									0.3	9	0.0	60
E 100											0.2	14

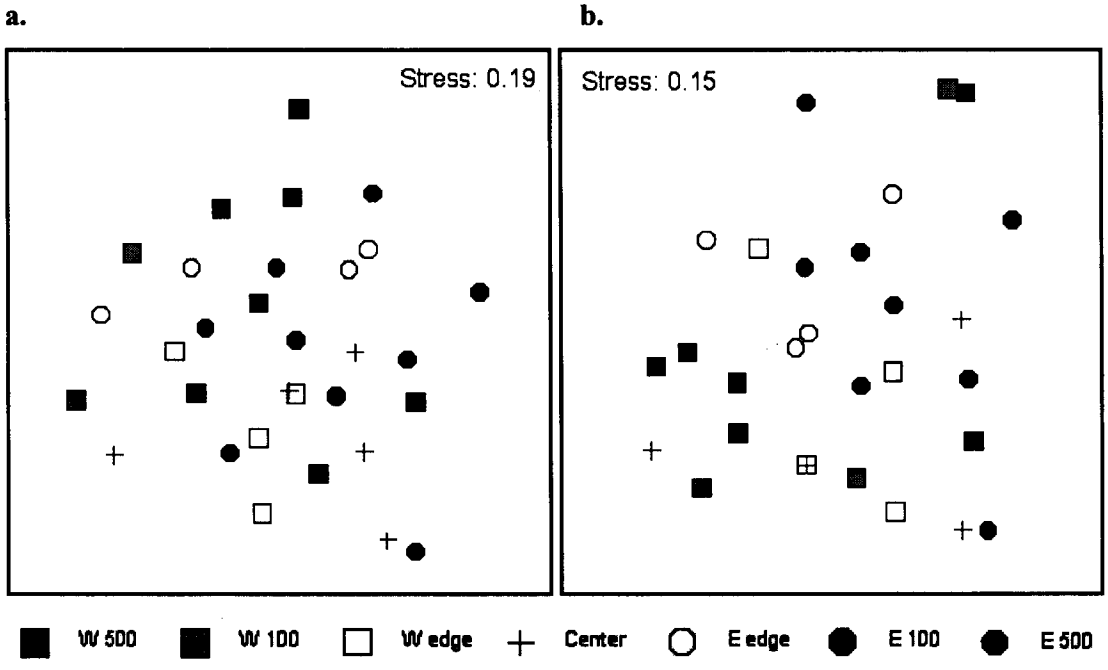


Figure 3.35 Two-dimensional nMDS of spring infauna assemblages from seven sites situated along transects running west and east from a low-density mussel-line aquaculture. MDS is based on Bray-Curtis dissimilarity matrix calculated from **(a)** square root transformed data, **(b)** presence/absence transformed data.

3.9.2 Faunal discriminators of benthic macroinfaunal assemblages: spring

Simper analysis of infauna data for spring shows that few taxa distinguished the sites from each other, reflecting the few significant pairwise comparisons. The East 100 m site is distinguished from the W 500 m site and the Centre by high abundances of capitellids (Table 3.27). As in summer, capitellids were absent from the West 500 m site. The Centre is distinguished from the East edge site by high abundance of ischerochids (Table 3.27).

Table 3.27 Simper results of data from spring infauna, from seven sites along transects, running west and east from a low-density mussel-line aquaculture farm. The Simper results were calculated from square root transformed data. Note: Only the species with "average dissimilarity divided by standard deviation" above 1.4 are included.

Sites		Site Av. Diss.	Family	Av. Abund.	Av. Abund.	Av. Diss.	Diss. / SD	Contrib. %
W 500	vs E 100	89	Capitellidae	0.0	0.8	7.2	1.4	8
Centre	vs E edge	92	Ischerochidae	1.2	0.0	12.6	1.5	14
	vs E 100	85	Capitellidae	0.0	0.8	7.7	1.4	9

3.9.3 Environmental determinants of benthic macroinfaunal assemblage patterns: spring

BVSTEP selected percentage cover of *A. antarctica*, total nitrogen in the water-column (bottom), seagrass leaf extension, chlorophyll-a in the water-column (bottom), seagrass leaf maximum height, and the ratio of epiphyte calcium carbonate to seagrass biomass, as the best combination of variables "explaining" the pattern of composition for infauna in spring (Table 3.28). When the composition of infauna, was limited to the most contributing taxa (equating to 95% of the pattern), BVSTEP selected *A. antarctica* %cover, water-column bottom total nitrogen, sediment loss on ignition, water-column bottom chlorophyll-a, seagrass leaf extension, and the ratio of epiphyte calcium carbonate to seagrass biomass, as the best combination (Table 3.29) accounting for the pattern in assemblages of the most contributing taxa. When the composition of infauna was limited (using Simper) to taxa responsible for the most significant differences among sites, BVSTEP selected water-column bottom nitrate/nitrite, epiphyte biomass, macroalgae % cover, as the best combination (Table 3.30) accounting for the pattern in taxa composition, based on Simper selected infauna, in spring.

In summary, *A. antartica* % cover, chlorophyll-a in the water-column (bottom), total nitrogen in the water-column (bottom), seagrass leaf extension, the ratio of epiphyte calcium carbonate to seagrass biomass (Table 3.28 & 3.29), sediment loss on ignition (Table 3.29), epiphyte biomass, macroalgae %cover, total nitrogen in the water-column (surface), and nitrate/nitrite water-column (bottom) (Table 3.30) are shown to be important for "explaining" the pattern in the composition of infauna at sites along the transect at the Misery Beach mussel farm in spring.

Table 3.28 Seagrass ecosystem attributes ranked using BVSTEP according to individual correlation to the taxa composition of infauna, in spring. Note: Variables forming the most significant (P<0.5) BVSTEP combination are highlighted.

1 %cover *Amphibolis antarica*	**0.645**	19 Seagrass dry mass	-0.012
2 Total nitrogen (bottom)	**0.592**	20 Nitrite / nitrate (bottom)	-0.016
3 %cover *Posidonia sinuosa*	0.553	21 Epiphyte dry / seagrass dry mass	-0.018
4 %cover *Halophila ovalis*	0.537	22 Leaf density	-0.025
5 %cover macroalgea	0.520	23 Total nitrogen (surface)	-0.030
6 Leaf extention (mm per day)	**0.420**	24 Epiphyte dry mass	-0.048
7 Chlorophyll 'a' (bottom)	**0.396**	25 NH4 (bottom)	-0.083
8 %cover bare sand	0.351	26 Chl 'a' sediment	-0.084
9 Loss on ignition from sediment	0.349	27 Ashfree epiphyte mass	-0.097
10 Shoot density	0.186	28 Depth	-0.104
11 NH4 in porewater	0.159	29 DIN water (bottom)	-0.112
12 Leaf maximum height	**0.114**	30 CaCO3 epiphyte / ashfree epiphyte	-0.153
13 CaCO3 epiphyte mass	0.082	31 Ashfree epiphyte / seagrass mass	-0.158
14 CaCO3 epiphyte / seagrass dry mass	**0.075**	32 Light attenuation	-0.171
15 Leaf Area Index	0.055	33 Chlorophyll 'a' (surface)	-0.215
16 Leaf extention (mg per day)	0.052	34 NH4 (surface)	-0.276
17 DIN porewater	0.000	35 Dissolved inorganic nitrogen (surface)	-0.296
18 Nitrite / nitrate (porewater)	-0.007	36 Nitrate / nitrate (surface)	-0.393
BV Step Combination of variables 1, 2,	**6, 7, 12, 14**	**Rho:**	**0.916**

Table 3.29 Seagrass ecosystem attributes ranked using BVSTEP according to individual correlations to the taxa composition of the infauna taxa contributing 95% of the MDS pattern, in spring. Note: Variables forming the most significant (P<0.5) BVSTEP combination are highlighted. Variables with significant individual correlation are denoted with an asterisk (*).

1 %cover *Amphibolis antarica*	**0.726**	19 Leaf density	-0.014
2 Total nitrogen (bottom)	**0.672**	20 Nitrite/nitrate (porewater)	-0.018
3 %cover *Halophila ovalis*	0.631	21 Seagrass biomass	-0.021
4 %cover macroalgea	0.530	22 Ashfree epiphyte biomass	-0.049
5 %cover *Posidonia sinuosa*	0.463	23 Total nitrogen (surface)	-0.059
6 Loss on ignition from sediment	**0.454**	24 Chl-a sediment	-0.061
7 Chlorophyll-a (bottom)	**0.344**	25 CaCO3 epiphyte/ashfree epiphyte	-0.065
8 %cover bare sand	0.212	26 Epiphyte biomass	-0.068
9 Leaf extention (mm per day)	**0.195**	27 DIN porewater	-0.069
10 NH_4 in porewater	0.149	28 Ashfree epiphyte/seagrass mass	-0.109
11 CaCO3 epiphyte/seagrass dry mass	**0.086**	29 Nitrite / nitrate (bottom)	-0.129
12 Shoot density	0.074	30 Leaf extention (mg per day)	-0.141
13 Leaf maximum height	0.070	31 NH_4 (surface)	-0.142
14 NH4 (bottom)	0.061	32 Chlorophyll-a (surface)	-0.144
15 Leaf Area Index	0.060	33 Light attenuation	-0.158
16 $CaCO_3$ epiphyte biomass	0.053	34 Dissolved inorganic nitrogen (surface)	-0.174
17 DIN water (bottom)	0.033	35 Nitrate/nitrate (surface)	-0.260
18 Epiphyte/seagrass biomass	0.021	36 Depth	-0.288
BV Step Combination of variables:	**1,2, 6, 7, 9,11**	**Rho:**	**0.871**

Table 3.30 Seagrass ecosystem attributes ranked according to individual correlations to the composition of the most important benthic infauna taxa distinguished by Simper analysis, in spring. Note: Variables forming the most significant (P<0.5) BVSTEP combination are highlighted. Variables with significant individual correlation are denoted with an asterisk (*).

1 **Nitrite/nitrate (bottom)**	**0.556**	19 %cover bare sand	0.090
2 Epiphyte/seagrass biomass	0.469	20 Ashfree epiphyte/seagrass biomass	0.038
3 Leaf extention (mm per day)	0.430	21 Total nitrogen (bottom)	0.018
4 Leaf extention (mg per day)	0.353	**22** Leaf density	-0.031
5 Total nitrogen (surface)	**0.348**	23 Chlorophyll-a in the sediment	-0.056
6 Epiphyte biomass	**0.289**	24 DIN porewater	-0.065
7 %cover *Amphibolis antarica*	0.239	25 CaCO3 epiphyte/seagrass biomass	-0.072
8 %cover macroalgea	**0.217**	26 Chlorophyll-a (surface)	-0.096
9 Nitrite/nitrate (porewater)	0.211	27 Chlorophyll-a (bottom)	-0.120
10 Loss on ignition from the sediment	0.195	28 CaCO3 epiphyte biomass	-0.137
11 NH_4 (bottom)	0.194	29 NH4 in porewater	-0.152
12 %cover *Posidonia sinuosa*	0.194	30 Shoot density	-0.162
13 Dissolved inorganic nitrogen (surface)	0.170	31 Ashfree epiphyte mass	-0.198
14 DIN water (bottom)	0.164	32 CaCO3 epiphyte/ashfree epiphyte biomass	-0.201
15 %cover *Halophila ovalis*	0.158	33 Depth	-0.222
16 NH4 (surface)	0.155	34 Nitrate/nitrate (surface)	-0.247
17 Light attenuation	0.151	35 Leaf Area Index	-0.261
18 Leaf maximum height	0.116	36 Seagrass biomass	-0.271
BV Step Combination of variables:	**1, 5, 6, 8**	**Rho:**	**0.881**

CHAPTER 4: DISCUSSION

This study has demonstrated differences in macrofauna assemblage structure, seagrass and epiphyte characteristics, and sediment ammonium concentrations among sites of varying distances from a mussel farm that was established in 1994. It provides correlative evidence that some components of the seagrass ecosystem varied according to proximity to mussel-line aquaculture at Misery Beach. This discussion will initially consider results for the summer season, as this season generally represents the typical peak of productivity for most temperate marine ecosystems (Barnes 1993, Cambridge & Hocking 1997, Hemminga & Duarte 2000, Short & Coles 2001) because of the light availability and temperature that are more conducive to plant growth (Carter 1991). Results for spring are discussed in section 4.3 SEASONAL VARIABILTY. The effects of shellfish aquaculture on benthic environments are also most apparent at this time of year (Kaspar et al. 1985, Kautsky & Evans 1987, Grant et al. 1995, De Casabianca et al. 1997b, Yokoyama 2002). The temporal consistency of trends at Misery Beach will be examined by comparing summer and spring results. This discussion firstly examines changes in benthic macrofauna composition and subsequently, organic enrichment, benthic vegetation, and seasonal variability. Pathways of cause and effect of mussel aquaculture on a seagrass ecosystem are discussed, and the discussion finally provides management implications of the findings from the study.

4.1 BENTHIC FLORA

Reduced seagrass biomass, leaf density and leaf area index at the East 500 m site, and low ratios of epiphyte biomass to seagrass biomass at the West 500 m site, were not considered to be clear evidence of an effect of mussel aquaculture on benthic flora at Misery Beach, and no significant differences among sites were recorded for seagrass leaf extension rates, maximum leaf height, and epiphyte ashfree biomass. However, epiphyte biomasses and epiphyte $CaCO_3$ biomasses were lower at both the West and East 500 m sites than at the other sites, and shoot densities were higher at both the West and East 500 m sites than at other sites.

No clear relationships were evident between light-attenuation and proximity to the mussel farm, nor between benthic flora and light. Structures associated with many forms of aquaculture potentially block light to the benthos (Delgado et al. 1997, Mendez et al. 1997), however, light levels below the farm were well above lower limits for *P.sinousa* (Kirkman & Kuo 1990, Duarte 1991, Masini & Manning 1997). This is consistent with most other studies of mussel-line aquaculture impacts, which have not reported reductions in light. Contrary to the findings of

Olsson & Graneli (1992) and Prins *et al.* (1996), mussel aquaculture at Misery Beach did not appear to improve light quality through mussels filtering the water and altering phytoplankton biomass. Nevertheless, water clarity was relatively high at all sites in Misery Beach, and a high degree of mixing contributed to the relative homogeneity of the water. This is characteristic of King George Sound, where *P. sinuosa* has a depth limit of at least 17 m, as it is well flushed and oligotrophic (Kirkman and Kuo 1990).

Seagrass biomass, leaf density and leaf area index, were not significantly different between most sites, however the variables showed trends, which were not statistically significant, but did reflect an increase close to the farm. Decreasing seagrass shoot densities at sites towards the farm opposed the trends for other seagrass variables. Reduction in seagrass shoot densities is known to be one of the primary and most evident responses to reduced light (Ruiz & Romero 2003) particularly that caused by increased epiphytes (Wood & Lavery 2000, Ruiz & Romero 2001, Hauxwell et al. 2003). Shoot density is considered to respond negatively to higher levels of nutrients and in indirect proportion to epiphyte biomass; with low shoot density and high epiphyte biomass typically occurring in eutrophic communities (Lapointe *et al.* 1994). Given their sensitivity to shading, low shoot densities at sites beneath and adjacent the Misery Beach mussel farm, could be explained by high epiphytes biomasses, stimulated by elevated ammonium fluxes. Although LOI did not provide evidence of organic enrichment, concentrations of ammonium in the porewater were greatest beneath the farm and could be linked to biodeposition from mussel aquaculture (Tenore et al. 1982, Mattsson & Linden 1983, Kaspar et al. 1985, Kautsky & Evans 1987, Baudinet 1990, Grenz et al. 1990, Hatcher et al. 1994, La Rosa et al. 2000, Chamberlain et al. 2001, Stenton-Dozey et al. 2001).

Major reductions in seagrass shoot densities and loss of seagrass due to finfish aquaculture have been reported in several studies on the impact of finfish farms (Mendez et al. 1997, Delgado et al. 1999, Pergent et al. 1999, Dimech et al. 2000, Ruiz et al. 2001). Thus, reduced shoot densities are potentially related to mussel aquaculture, via nutrient enrichment and reduction in seagrass photosynthesis. Bioavailable nutrients, such as ammonium, could have effluxed from the sediment (Asmus & Asmus 1991, Ogilvie et al. 2000, Peterson & Heck Jr 2001a) where elevated concentrations may have pooled below the seagrass canopy, promoting epiphyte growth. Decreased top-down control by unrecorded herbivorous fish beneath and adjacent the farm, may also have contributed to increased epiphyte biomasses (Sala & Boudouresque 1997, Ruitton et al. 2000).

Unlike other studies that have reported reduced seagrass biomasses, leaf areas and leaf extension rates at sites close to finfish aquaculture (Mendez et al. 1997, Delgado et al. 1999, Pergent et al.

CHAPTER 4: DISCUSSION

This study has demonstrated differences in macrofauna assemblage structure, seagrass and epiphyte characteristics, and sediment ammonium concentrations among sites of varying distances from a mussel farm that was established in 1994. It provides correlative evidence that some components of the seagrass ecosystem varied according to proximity to mussel-line aquaculture at Misery Beach. This discussion will initially consider results for the summer season, as this season generally represents the typical peak of productivity for most temperate marine ecosystems (Barnes 1993, Cambridge & Hocking 1997, Hemminga & Duarte 2000, Short & Coles 2001) because of the light availability and temperature that are more conducive to plant growth (Carter 1991). Results for spring are discussed in section 4.3 SEASONAL VARIABILTY. The effects of shellfish aquaculture on benthic environments are also most apparent at this time of year (Kaspar et al. 1985, Kautsky & Evans 1987, Grant et al. 1995, De Casabianca et al. 1997b, Yokoyama 2002). The temporal consistency of trends at Misery Beach will be examined by comparing summer and spring results. This discussion firstly examines changes in benthic macrofauna composition and subsequently, organic enrichment, benthic vegetation, and seasonal variability. Pathways of cause and effect of mussel aquaculture on a seagrass ecosystem are discussed, and the discussion finally provides management implications of the findings from the study.

4.1 BENTHIC FLORA

Reduced seagrass biomass, leaf density and leaf area index at the East 500 m site, and low ratios of epiphyte biomass to seagrass biomass at the West 500 m site, were not considered to be clear evidence of an effect of mussel aquaculture on benthic flora at Misery Beach, and no significant differences among sites were recorded for seagrass leaf extension rates, maximum leaf height, and epiphyte ashfree biomass. However, epiphyte biomasses and epiphyte $CaCO_3$ biomasses were lower at both the West and East 500 m sites than at the other sites, and shoot densities were higher at both the West and East 500 m sites than at other sites.

No clear relationships were evident between light-attenuation and proximity to the mussel farm, nor between benthic flora and light. Structures associated with many forms of aquaculture potentially block light to the benthos (Delgado et al. 1997, Mendez et al. 1997), however, light levels below the farm were well above lower limits for *P.sinousa* (Kirkman & Kuo 1990, Duarte 1991, Masini & Manning 1997). This is consistent with most other studies of mussel-line aquaculture impacts, which have not reported reductions in light. Contrary to the findings of

1999, Ruiz & Romero 2001), seagrass biomasses, leaf densities and leaf area indices at sites close to the Misery Beach mussel farm were apparently unaffected by higher biomasses of epiphytes. *Posidonia sinuosa* at Misery Beach could have been nutrient limited, thus elevated ammonium concentrations within the rhizosphere may have partially counteracted the effects of epiphyte shading by promoting seagrass productivity (Reusch & Williams 1998, Peterson & Heck Jr 2001b).

4.1.1 Organic enrichment and benthic flora

Although there was no chemical evidence that organic enrichment was responsible for differences in benthic flora, the trends for seagrass shoot densities and epiphyte biomasses appear to be related to the trend for of ammonium in the porewater, being greatest near the farm, which does suggest some subtle enrichment link. Variations in organic content of the sediment were not detected, possibly due to the organics being resuspended by the strong water currents (Prins et al. 1996) prior to sampling, late summer. Alternatively, organic content could have been consumed seasonally by organisms, such as bacteria (Grenz et al. 1990, Lopez 1998), which were not measured in the study. If this was the case, bacteria potentially consumed and depleted oxygen in the sediment. However, this process could have been counteracted by the exudation of oxygen from seagrass roots (Short 1986, Filskov 1990, Holmer 1992, Martinova 1993, Hemminga 1998, Connell & Walker 2001).

Posidonia sinuosa meadows could have facilitated the retention and microbial decomposition of mussel biodeposition, resulting in mineralisation and increased ammonium concentrations at the benthos. However, built-up seagrass detritus could have obscured less substantial contributions of mussel biodeposited carbon to the carbon content of the sediment, and may explain why LOI was not different between sites in summer. *Posidonia* seagrass meadows are known to produce a net organic carbon accumulation in the sediments through the burial of seagrass detritus (Cambridge & Hocking 1997, McMahon 1998, Garcia et al. 2002), which has a low ratio of nitrogen to carbon (Garcia et al. 2002). Therefore seagrass-derived carbon was likely to have outweighed mussel derived carbon in the sediment at Misery Beach.

4.2 BENTHIC MACROFAUNA

The composition of benthic macrofauna at the site beneath the centre of the Misery Beach mussel farm was unlike any other site, and similarly, sites away from the farm had distinct taxa assemblages. In contrast, assemblages at sites on the edge and adjacent to the farm were similar to each other, but dissimilar to sites away from the farm. These patterns differed and were much more subtle than the findings of Mattson & Linden (1983) who reported pronounced impoverishment of benthic macrofauna assemblages close to a mussel aquaculture, but like that study and others (Kaspar et al. 1985, Chamberlain et al. 2001, Stenton-Dozey et al. 2001), this study found that mussel aquaculture influenced the community structure of benthic macrofauna. At Misery Beach, taxa richness of motile epibenthic macrofauna (epifauna) was high at sites adjacent to the farm relative to sites further away, suggesting a link between epifauna taxa composition and the close proximity of the farm. However, abundance was the main driving force of patterns in assemblage structure for epifauna. Although most epifauna taxa were recorded at all sites, total abundances of epifauna were highest at sites adjacent to the farm, and low at sites away from the farm and beneath the farm.

Patterns observed in the nMDS ordinations of benthic infauna assemblages were similar to those for epifauna, however, taxa richness contributed strongly to these patterns rather than abundance. Total taxonomic richness and densities of infauna were relatively high at sites adjacent to the farm, but lower at the West 500 m site, and extremely low at sites beneath the farm. This pattern was similar to those documented in many studies that have examined infauna along a gradient of organic enrichment (Pearson & Rosenburg 1978, Ritz et al. 1989, Weston 1990, Kroncke 1996, Carrasco & Carbajal 1998, Cardell et al. 1999, Samuelson 2001, Rossi 2002), including infauna in the vicinity of mussel aquaculture (Chamberlain et al. 2001).

Spatial changes in both epifauna and infauna were similar, and these patterns were consistent with macrofaunal responses associated with colonies of *Mytilus edulis* and mussel aquacultures in other parts of the world, where such patterns have been attributed to anoxia and predation (Castell et al. 1989), changes in food chains (Tenore et al. 1982), filtration of phytoplankton and other suspended particles, the biodeposition of faecal pellets, which leads to changes in habitat quality (Grenz et al. 1990, Ragnarsson & Raffaelli 1999).

4.2.1 Indicator taxa

In summer, amphipods dominated the epifauna taxa lists for all sites at Misery Beach, and most of the taxa driving the patterns in assemblages of epifauna were amphipods. Ochlesidae, Eusiridae, Protellidae and Dexaminidae were important taxa distinguishing the assemblages at sites adjacent to the farm, where they were relatively abundant. Ecological information on these taxa is limited, though amphipod densities have been known to be influenced by differences in habitat structure and food availability (Jernakoff & Nielsen 1997b, Bologna & Heck 1999, Aikins & Kikuchi 2001, Nakaoka et al. 2001). Most species of amphipods are scavengers and feed on detritus (Barnes 1987), while some are selective, deposit feeders (Hutchings et al. 1991) and others graze on seagrass and epiphytes (Sergeev et al. 1988).

With respect to infauna, the families Syllidae, Capitellidae and Spoinidae were important in distinguishing sites adjacent to the farm at Misery Beach. Trends for densities of Syllidae, Capitellidae and Spoinidae were representative of the general pattern seen for the total abundance of infauna: highest densities were at the site 100 m west, adjacent the farm; low densities were found away from the farm; and these taxa were absent beneath the farm. Most species of Syllidae, Capitellidae and Spoinidae are deposit feeding detritivores or feed on diatoms (Beesley et al. 2000). They generally have a preference for fine sediments and are tolerant of sediments with increased organic content (Pearson & Rosenburg 1978, Kroncke 1996), commonly associated with benthic environments below mussel aquaculture (Castell et al. 1989, Fabiano et al. 1994, Barranguet 1997, Mirto et al. 2000).

Polychaete communities are known to be influenced by differences in habitat structural complexity (Wilson 1981, Snelgrove & Butman 1994, Gambi et al. 1998, Webster 1998, Cardell et al. 1999). Opportunistic species, such as those from the Capitellidae family, have been shown to proliferate in response to newly vacated habitats associated with the impact of mussel aquaculture (Tenore et al. 1982, Mattsson & Linden 1983), whereas other taxa, such as some Syllidae species, have narrow habitat preferences, requiring relatively high seagrass density and sediments with increased silt-clay fraction (Beesley et al. 2000). Given patterns in fauna distribution and abundance, and what is known of the factors that can influence these patterns, it is likely that either organic enrichment, habitat structure, food availability or a combination of these, influenced the patterns observed at the Misery Beach site. The association between these variables and benthic macrofauna is explored in the following two sections.

4.2.2 Organic enrichment and benthic macrofauna

Although there was no chemical evidence that organic enrichment was responsible for differences in the benthic macrofauna community, the pattern of macrofauna distribution does suggest some longer-term organic enrichment link. Impacts of mussel aquaculture on local environments are usually due to the deposition of faeces, pseudo-faeces, gametes, whole mussels, fowling organisms and other suspended particles (Tenore et al. 1982, Kautsky & Evans 1987, Stenton-Dozey et al. 2001), which are known to lead to high organic content, hypoxia and decreased particle size in the sediment (Tenore et al. 1982, Mattsson & Linden 1983). These direct effects have been linked to dramatic shifts in benthic macrofaunal assemblages (Tenore et al. 1982, Mattsson & Linden 1983, Castell et al. 1989, Ragnarsson & Raffaelli 1999). In summer, loss on ignition of the sediments at Misery Beach did not differ among sites, however the trend for ammonium in the porewater suggests nitrogen enrichment of the benthos at sites beneath and adjacent to the farm. This pattern in porewater ammonium may be a result of both faecal deposition and the deposition of whole mussels, the shells of which were common, immediately beneath the farm (personal observation).

Water currents at Misery Beach flow westward in summer and are known to predominantly flow westward with the prevailing wind in the months prior to summer (Meteorology 1999, D'Adamo 2000). Based on the formulation of Chamberlain et al. (2001), and using average current velocity (*S*: 0.136 m s^{-1}) and direction (westerly, a: 270°), together with faecal settling rates (<0.5cm s^{-1} for faeces and <0.8 cm s^{-1} for pseudofaeces) and depth (12m), the majority of faecal and pseudo-faecal material are estimated to have deposited on the benthos between 75 and 235m west of its origin, which covers the farm itself and areas to the west. Therefore it is possible that the Centre, West edge and West 100 m sites were subjected to faecal or phytoplankton material in summer. However, the possibility of sites also being subjected to dislodged mussels or fowlers was restricted to the sites beneath the farm (Gibbs et al. 1992, Hatcher et al. 1994, La Rosa et al. 2000, and Chamberlain et al. 2001).

In addition to the biodeposition generated by the mussels, mussel aquaculture structures are also known to increase settlement of normally occurring suspended solids, such as phytoplankton, which contribute to organic sedimentation (Kautsky & Evans 1987, Hall et al. 1990, Gibbs et al. 1992, Hatcher et al. 1994, La Rosa et al. 2000), may help to explain the high biomass of epiphytes near the farm. This equates to increased fluxes of gametes, spores and larvae (Hovel et al. 2002), which can influence the benthic assemblages (Kautsky & Evans 1987, Ragnarsson & Raffaelli 1999, Chamberlain et al. 2001) including epiphytic algae and macrofauna.

Chamberlain et al. (2001) suggest that mussel aquaculture could potentially provide a food source to benthic macrofauna, and alter the characteristics of the benthos, by elevating the sedimentation of suspended particles, such as phytoplankton. It is therefore possible that the farm increased the settling rate of particles in the water column, other than mussel biodeposition, and could have influenced infaunal assemblage structures among sites near the Misery Beach mussel farm.

Assuming that areas in the immediate vicinity of the Misery Beach mussel farm were receiving enhanced biodeposition, particular epifaunal and infaunal taxa would potentially be favoured by enhanced food supplies or reproductive advantages (Radziejewska 1986, Kautsky & Evans 1987, Castell et al. 1989, Fabiano et al. 1994, Hargrave et al. 1997, Snelgrove et al. 2000b). In this study, BVSTEP showed a correlation between ammonium in the porewater and the pattern for epifauna assemblages. If the effects biodeposition from the mussel farm was linked to the high porewater ammonium concentrations, this may explain the correlation between porewater ammonium and epifauna assemblages. Grant et al. (1995) found that the impact of a mussel farm on benthic community structure was minor and mainly due to the impact of dislodged mussels, and it is possible that a similar influence was occurring at this study site. Possibly more significant is the influence of faecal material and finer waste from the mussel farm, which is known to be a high quality food source to particular epifauna taxa (Lee 1997, Cruz-Rivera & Hay 2000), due to its high C:N ratio (Kautsky & Evans 1987).

Many infauna taxa, including Syllidae, Capitellidae and Spoinidae, are known to proliferate in response to newly altered environments associated with biodeposition from mussel aquaculture (Tenore et al. 1982, Mattsson & Linden 1983, Hatcher et al. 1994). Syllidae, Capitellidae and Spoinidae were highest in abundance at the site 100 m west, adjacent to the farm, and lowest in abundance away from the farm. However, many infauna taxa, such as Eunicid and Paronid polychaetes, are less abundant in organically enriched sediments (Ragnarsson & Raffaelli 1999). At Misery Beach, Eunicid and Paronid polychaetes were most abundant away from the farm, while Paronid polychaetes were in low abundance and Eunicid polychaetes were absent beneath the farm. These families are commonly associated with non-polluted environments (Beesley et al. 2000), suggesting that conditions beneath the farm may have been unfavourable to species of Eunicidae, Paronidae, and Polynoidae due to habitat alterations.

It is possible that oxygen deficiencies in the sediment, related to organic enrichment, lead to mortality or migration of macrofauna that are intolerant to low levels of oxygen, resulting in decreased taxa diversity and total abundance (Pearson & Rosenburg 1978, Mattsson & Linden 1983, Warwick et al. 1987, Weston 1990). However, many species of Syllidae, Capitellidae and

Spoinidae have a tolerance of sediments with a high bacterial oxygen demand (Pearson & Rosenburg 1978, Kroncke 1996).While the study found no direct evidence of significant anoxia around the farm, the greatest abundance of these taxa, known to be relatively tolerant to hypoxia, were recorded at the West 100m site, which is potentially exposed to biodeposition of faeces and gametes during summer. Since differences in the composition of benthic infauna appear to have been driven by taxa richness as well as abundance, migration or mortality of taxa may explain these differences.

The presence of seagrass at Misery Beach is an important difference setting the present study apart from other studies of mussel farm impacts. At Misery Beach, the typical impacts from mussel aquacultures, such as organic enrichment, hypoxia, and dramatic shifts in the structure of benthic communities were not evident. Since *Posidonia sinuosa* was dominant at all sites and the water column was well mixed, hypoxia was unlikely, but the presence of hypoxic pockets of sediment, was possible. Organic enrichment potentially had an indirect effect on benthic macrofauna by influencing seagrass and epiphytes, equating to alterations in habitat structure or food availability.

4.2.3 Habitat structure and food availability

While trends in macrofauna assemblage structure point to an indirect association with porewater ammonium, they also correspond to patterns in seagrass and epiphytes. At the West and East 500m sites, abundances and total richness of epifauna, and epiphyte biomasses were lowest, while and seagrass shoot densities were highest. Composition of macrofauna is often affected by changes in habitat structure (Wilson 1981, Edgar & Robertson 1992, Martin-Smith 1993, Jernakoff & Nielsen 1997b, Gambi et al. 1998, Sanchez-Jerez et al. 2000, Hovel et al. 2002) and correlations between vegetation biomass and macrofauna densities and diversity are common (Bell & Westoby 1986, Sogard et al. 1987, Sergeev et al. 1988, Ferrell & Bell 1991, Edgar & Robertson 1992, Valentine & Heck 1993). The habitat preferences of individual species of epifauna are established by complex relationships between each species' requirement for food and shelter (Sergeev *et al.* 1988), and aspects of seagrass such as biomass and leaf area index, affect how well a patch will function to provide habitat and food. The species composition and abundances of epifauna and infauna in seagrass meadows are known to be influenced by features of meadow structure such as shoot density (Bell & Westoby 1986, Connolly & Butler 1996, Jernakoff & Nielsen 1997a, Gambi et al. 1998, Lee et al. 2001, Nakaoka et al. 2001, Hovel et al. 2002).

At Misery Beach in summer, three of the four most influential variables on the patterns in epibenthic macrofauna and infauna assemblages were related to habitat characteristics. BVSTEP analysis showed that epibenthic macrofauna composition was likely to be influenced by ammonium in the porewater, the maximum height of seagrass leaves, percentage cover of *Amphibolis antarctica* and the ratio of ashfree epiphyte to seagrass biomass. At the West and East 500m sites, abundances and total richness of epifauna were lowest, where ammonium in the porewater was also lowest. The seagrass and epiphyte variables identified by BVSTEP as being influential to epifauna, did not vary significantly among sites, however, they could be linked to porewater ammonium, at some magnitude. Therefore, the BVSTEP result supports the possibility that porewater ammonium concentrations indirectly affected epifauna taxa composition, through an effect on habitat and food resources provided by seagrass and epiphytes.

In this study, it is difficult to disassociate seagrass from epiphytes in relation to their influence on macrofauna, because seagrass and epiphytes are likely to have influenced each other. Bologna & Heck (1999) suggest that epifaunal assemblages associated with seagrass are dramatically influenced by the trophic function of epiphytes, consistent with Jernakoff and Nielson (1997) who suggested that amphipods are active selectors of the epiphyte taxa they graze on. For grazing epifauna, high epiphyte biomass at sites near the Misery Beach mussel farm, could have equated to increased food availability (Hall & Bell 1993). Differences in seagrass structure, such as leaf area indices and shoot densities potentially influenced the area of substrate available to epiphytes, thereby altering epiphytes assemblages. Given the known specificity in food choices even among related species of epifauna (Nakaoka et al. 2001), differences in seagrass structure could indirectly influence epifaunal composition (Robertson & Mann 1980, Barnes 1987, Edgar 2000, Jones & Morgan 2002), at Misery Beach, through their influence on particular epiphytic algal species (Bell & Westoby 1987, Parker et al. 2001). Equally, these parameters can relate to habitat structural complexity. High epiphyte biomass and seagrass leaf area indices at sites adjacent to the farm at Misery Beach potentially inhibited top down pressure, which could explain high taxa richness and densities of epifauna and infauna adjacent to the farm. Amphipods tend to prefer habitats with high complexity, which is known to be important in structuring epifauna assemblages (Aikins & Kikuchi 2001). High seagrass leaf area and epiphyte biomass could have provided higher complexity, and potentially mitigated the effects of predation on benthic macrofauna by providing refuges for prey (Orth et al. 1984, Bell & Westoby 1987, Hall & Bell 1988, Schneider & Mann 1991).

The only anomaly in the apparent correlation between habitat structure and epibenthic faunal assemblage was at the site immediately below the farm. Here, seagrass leaf area indices and

epiphyte biomass were high, compared to sites away from the farm, yet epifauna and infauna were at lowest abundances. Macrofaunal assemblage structures were apparently influenced by changes in the seagrass and epiphytes adjacent to the farm, however, it is also possible that the environmental factors that influenced the flora beneath the farm, had an equal or greater direct influence on associated macrofauna assemblages, than influence of the flora itself (Snelgrove et al. 2000a). Thus, while this study cannot clarify the relative importance of seagrass or epiphytes, or shelter versus food provision roles, it is clear the structure of epibenthic assemblages was correlated to habitat features.

Epiphyte biomasses were greatest at sites beneath and adjacent to the mussel farm. Since epifauna are known to be epiphyte grazers (Sergeev et al. 1988), or use epiphytes as refuge from predators (Martin-Smith 1993), epiphyte biomass could help explain differences in epifaunal taxa composition. While this might explain the observed faunal pattern at Misery Beach, the pattern in epiphytes contrasts with those found elsewhere, where lower epiphyte biomasses have been reported (Peterson & Heck Jr 2001a, & b) and beneath aquaculture (Ruiz *et al.* 2001). Peterson & Heck Jr (2001a) suggested that epiphytes were reduced by epibenthic grazers, beneath a mussel farm in the Gulf of Mexico, due to an increase in their palatability. The high epiphyte biomasses beneath the farm at Misery Beach, compared to sites away from the farm may be explained in the same way if the high biomasses at these sites, largely resulted from calcareous epiphytes, which are less palatable to benthic grazers. If this were the case, it would also suggest that increased habitat provision, rather than increased food availability, is a more likely explanation for peak abundances of epifauna at sites down-current from the farm.

With respect to infauna, BVSTEP analysis shows that the ratio of calcium carbonate to seagrass dry mass was one of the most important variables influencing assemblage structure. However, it is important to note that this analysis is based on five sites only, and excluded the Centre and West edge site due to an absence of infauna in the samples. Nevertheless, it is plausible that infauna assemblages were influenced by the ratio of calcium carbonate to seagrass dry mass. A proportion of epiphyte debris is usually recycled into the sediment (Cambridge & Hocking 1997), where it forms fine mineral deposits. Since epiphyte calcium carbonate biomass was highest at sites near the farm, input of epiphyte calcium carbonate could have influenced the grain size at these sites. Although this study did not measure sediment grain size, it is possible that the particle size of sediments influenced infauna, such as deposit feeding polychaetes, as it relates to habitat and feeding modes (Cardell et al. 1999, Samuelson 2001), and may be an important variable to record in future studies of seagrass communities beneath mussel aquaculture .

A final habitat feature that may have influenced patterns in faunal assemblages is the abundance of MPB in the sediment. Microphytobenthos (MPB) is a potential food source for infauna (Barnes 1987, Beesley et al. 2000), and changes in MPB abundance due to aquaculture can be reflected in patterns for infauna (Mirto *et al.* 2000). At Misery Beach sediment chlorophyll-a concentrations were not significantly different between sites, though weak and insignificant trends among sites reflected observed patterns for infauna composition. Densities and taxa richness of epifauna and infauna, and concentrations of sediment chlorophyll-a, display peaks at the West 100 m site, which was differentiated from the East edge and the East 100 m sites by a high abundance of spoinid polychaetes. Rossi (2002) suggests that MPB could influence abundance of polychaetes, such as spoinids, by increasing the nutritional value of the sediment that some infauna ingest. The present study could not present a definitive variable associated with shifts in infauna taxa composition, however it seems likely that infauna assemblage structures were influenced by sediment characteristics that were determined by the relative quantities of seagrass detritus, mussel waste, plankton, gametes, spores, and larvae deposited to the benthos. These variables, including MPB, can directly influence infauna assemblage structures by providing food and/or reproductive advantage, or by affecting habitat quality (Kaspar et al. 1985, Castell et al. 1989, Snelgrove & Butman 1994, Kroncke 1996, Turner 1997, Carrasco & Carbajal 1998, Dial & Roughgarden 1998, Ragnarsson & Raffaelli 1999, Rossi 2002).

4.3 SEASONAL VARIABILITY

Similar to summer, species composition of epifauna and infauna in spring displayed differences among sites close to the farm and sites away from the farm. However, in contrast to summer, taxa richness and abundances of epifauna in spring were high at sites beneath, as well as adjacent to the farm. Sites away from the farm recorded epifauna densities that were low in both summer and spring. In terms of infauna, the highest densities were recorded east of the farm, which contrasted with the pattern found in summer. However, species richness was lowest at the Centre site for both seasons. Sediment chlorophyll-a and seagrass leaf extension rates were lower in spring than in summer, and most other seagrass and epiphyte variables showed significant interactions between site and seasons, and were generally greater in summer, depending on the site.

Seasonal changes in variables, such as water current direction and velocity, water temperature and light intensity, were probably responsible for many of the differences observed between

summer and spring. Water temperature was as low as 16 °c in spring, compared to 21 °c in summer. Similar to other regions, the effects of any organic enrichment on benthic communities were therefore less likely to be apparent in spring (Kaspar et al. 1985, Kautsky & Evans 1987, Grant et al. 1995, De Casabianca et al. 1997a, Yokoyama 2002).

Mussel activity is normally highest during summer, which is the same season, in which nutrients can limit the primary productivity of seagrass and epiphytes (Kautsky & Evans 1987) due to peak productivity. Assuming mussel aquaculture was a source of nutrients in summer, any nutrient enrichment to the seagrass communities could stimulate higher seagrass and epiphyte biomass (Reusch et al. 1994, Reusch & Williams 1998, Peterson & Heck Jr 1999, 2001a, b) through elevation of nutrient limitation. However, during spring, the seagrass and epiphytes were potentially limited by light and temperature. Epiphyte biomasses at the sites away from the farm during spring and summer were similar, yet values for epiphytes at the West and East edge sites and the West 100 m site were much lower in spring than in summer. this pattern might be explained by high levels of porewater ammonium at sites close to the farm promoting epiphytes in summer, while the temperature were conducive to growth, however, during spring epiphyte growth, limited by light and temperature, could have been heavily grazed by the high abundances of epifauna observed at these sites. During summer, elevated epiphyte biomasses at sites near the farm, potentially reduced light to the bulk of the seagrass leaves through self shading, resulting in decreased shoot density at sites near the farm.

Water currents at Misery Beach flow eastward in spring and are known to predominantly flow eastward with the prevailing wind in the months prior to spring (Meteorology 1999, D'Adamo 2000). Based on the formulation of Chamberlain et al. (2001), and using average current velocity (S: 0.074 m s^{-1}) and direction (easterly, a: 83°), together with faecal settling rates (<0.5cm s^{-1} for faeces and <0.8 cm s^{-1} for pseudofaeces) and depth (12m), the majority of faecal and pseudo-faecal material is estimated to have deposited on the benthos between 40m and 135m east of the origin. Therefore, the Centre site potentially received biodeposition, including whole mussels, in both seasons. Sites immediately east of the farm (beneath and adjacent) were probably exposed to faecal deposition only during spring.

4.4 THE BENTHIC MODEL

This study provides evidence of a benthic pathway of cause and effect of mussel aquaculture on a seagrass ecosystem. Mussel aquaculture is known to result in biodeposition at the benthos in the form of faeces, pseudofaeces, gametes, dead mussel and fowling organisms (Kautsky & Evans 1987). Fallen mussel shells can provide some taxa of macrofauna with refuge from

predators, such as fin-fish, while, faecal waste, depositing at the benthos, can reduce the grain size of sediment habitats. Therefore organic waste could directly alter habitats of benthic macrofauna, and provide an available food source to particular guilds of benthic macrofauna, directly influencing faunal assemblage structures. This study demonstrates significant differences in the taxa composition among sites at increasing distances to a mussel aquaculture. However, mussel waste could indirectly affect benthic macrofaunal assemblage structures. As mussel waste is incorporated into the sediment it is colonised by microbes including denitrifying bacteria (Fabiano et al. 1994), which leads to remineralisation of nutrients and consumption of oxygen through respiration (Mazouni 1996). This study did not measure oxygen levels in the sediment; however any depletion of oxygen in the sediment could have contributed to stress on most benthic macrofauna taxa. Remineralised nutrients within the sediment porewater could facilitate changes in seagrass and microphytobenthos, thus affecting benthic macrofauna through altered habitat or through food provision. This study showed that concentrations of ammonium in the porewater were significantly higher at sites close to the farm, than away from the farm. Some of the remineralised nutrients in the sediments could flux to the water column through a benthic-pelagic pathway promoting epiphyte growth on seagrass leaves, thereby reducing the capacity of seagrass to photosynthesise. The trend for epiphytes resembled the trend for porewater ammonium, having significantly greater epiphyte biomass at sites close to the farm, than sites away. The trend for seagrass shoot densities corresponds inversely to both the trends for epiphyte biomasses and porewater ammonium. Differences in seagrass and epiphytes could have influenced seagrass associated macrofauna, through habitat alterations and changes in food availability. Taking into account seasonal differences, including changes in hydrodynamics, the results of this study support its initial benthic model.

4.5 MANAGEMENT IMPLICATIONS AND CONCLUSIONS

The present study at Misery Beach provides correlative evidence that assemblage structures of epifauna, infauna and the seagrass habitat were altered according to proximity of mussel-line aquaculture. In the introduction to this thesis, a number of potential cause-effect pathways were hypothesised, whereby mussel-line aquaculture could influence seagrass ecosystems. The most likely conceptual pathway of cause-effect from the mussel-line aquaculture to the seagrass community at Misery Beach essentially involves benthic variables; since changes in, light, chlorophyll-a and nutrients in the water column showed no clear relationships with other measured variables. With respect to macrofauna, the data are most consistent with an influence of aquaculture on seagrass structure flowing on to benthic macrofauna, through either trophic or

habitat processes. The mechanism of effect on seagrass and epiphytes, however, remains unclear. It appears likely that seagrass and epiphyte growth could have been influenced by ammonium accumulated in the porewater, following the microbial decomposition of mussel waste and detached whole mussels. Current velocities indicate that the degree of flushing was higher at Misery Beach than documented in other studies of mussel aquaculture impacts. It is therefore possible that this flushing reduced the impact on seagrass beneath the farm and that any biodeposition that occurred spread beyond the farm boundaries as well as beneath the farm, with current direction dependent on season. Although the present study provides no evidence of organic enrichment, the patterns of macrofauna distribution and ammonium concentrations at the benthos are consistent with some longer-term organic enrichment link.

The present study, like most benthic surveys, assesses potential gradients in disturbance using a limited number of sampling points, which does not give a complete picture of the differences in macrofauna assemblages (Pearson & Rosenburg 1978). However, a general trend of highest total abundances for epifauna and infauna at intermediate proximity (100m) to a potential source of disturbance (mussel aquaculture), is comparable to patterns described in the classic paper by Pearson and Rosenburg (1978). Using illustrative terminology, a 'step effect' was apparent, moving away from the Misery Beach mussel farm. Total abundances and taxa richness of infauna were lowest at sites beneath the farm, highest at the site 100 m west adjacent to the farm, and intermediate away from the farm, conforming to the intermediate disturbance principle (Connell 1978).

At the stocking density of approximately 1.3 tonnes of mussels per hectare, the effects of mussel-line aquaculture at Misery Beach on seagrass communities at an ecosystem scale were subtle compared to impacts reported in the literature. However, the study provides evidence of differences in seagrass communities, particularly epibenthic macrofauna and infauna, among sites within a hundred metres from the farm, and those further away. The findings leave open the possibility that mussel aquaculture at higher stocking densities could be associated with greater changes to surrounding seagrass communities. Maguire (2002) stated that past monitoring of impacts on seagrass beds beneath mussel-lines at Albany had indicated negligible effects. Although the influence of the mussel farm in the present study may also be minor, results apply to a low-density mussel farm that had operated for six years in a well flushed environment. Therefore, the results may not be representative of mussel farms that have; larger stocking densities, operated over longer periods of time, or existed within environments that are poorly flushed.

At Misery Beach, differences among sites were less dramatic than impacts described in most

other studies of mussel aquaculture in unvegetated environments, due to the low density of mussels on the farm, and/or the degree of flushing, and/or the presence of seagrass. The low density of mussels is likely to have equated to relatively low quantities of mussel waste, hence any possible effect of the farm on the ecosystem was limited. The degree of flushing is likely to have been responsible for the dispersal of mussel waste to the seagrass communities and therefore its concentration at the benthos (Mirto et al. 2000).

Posidonia sinuosa meadows could have facilitated the retention of depositing material, including mussel waste, plankton, gametes, spores, and larvae. Furthermore, leaked oxygen from seagrass roots could have counteracted the bacterial oxygen demand in the sediment (Filskov 1990, Hemminga 1998, Connell & Walker 2001), facilitating rapid decomposition of mussel waste, and oxidisation of faecal ammonium to form highly soluble nitrates (La Rosa et al. 2000). Seagrass ecosystems are commonly nutrient-poor and often nitrogen limited in the south-west of Western Australia (Cambridge & Hocking 1997, Udy 1997b, a, McMahon 1998) thus, at Misery Beach it is possible that most of the nutrients in the porewater derived from mussel waste were absorbed and utilised by the seagrass (Martinova 1993, Reusch et al. 1994, Peterson & Heck Jr 1999, Touchette 2000, Peterson & Heck Jr 2001a, b, Stenton-Dozey et al. 2001, Garcia et al. 2002). Seagrass processes at Misery Beach may have mitigated the effect of biodeposition and helped maintain chemical and physical properties of the sediment, thus inhibiting opportunist species of flora and fauna from totally out-competing less tolerant species, and reducing the diversity of the benthic infauna communities at all sites beneath and adjacent the farm.

Epifauna and infauna assemblages were the most sensitive indicators of changes at varying proximity to aquaculture. It is therefore recommended that benthic macrofaunal assemblages be used for monitoring the effects of aquaculture (Warwick 1988, Karakassis & Hatziyanni 2000). In particular Dexaminidae and Ochlesidae amphipods, Protellidae skeleton shrimps and polychaetes Capitellidae, Polynoidae, Spoinidae and Syllidae, were useful families for discriminating sites near the farm, from site away. Multivariate analysis proved a simple and valid method for identifying variations in taxa assemblages among sites, and relating these differences to environmental factors (Clarke 1993, Clarke & Warwick 1994).

Prior to the establishment of an aquaculture facility, physicochemical and biological properties of the waste of that type of aquaculture should be assessed in relation to the sensitivity of the local ecosystem components. As suggested by Chamberlain et al. (Chamberlain et al. 2001) mathematical models could be used to predict dispersal of waste products, allowing sampling points to be chosen accordingly. Finally, it is important that monitoring programs include a

summer sampling period. Spatial trends in ecosystem variables at Misery Beach were strongest in summer, and provide a pertinent example of seasonally dependent effects from mussel-line aquaculture on a seagrass ecosystem. Overall, the study documents an example of low-density mussel-line aquaculture over seagrass communities, where functions and values of the environment were sustained at an ecosystem scale, but on a local scale the study also provides evidence of a farm's seagrass influence on the seagrass communities, particularly epibenthic macrofauna and infauna, and suggests that mussel aquaculture at higher stocking densities, or/and in less flushed environments could cause greater changes to surrounding seagrass communities.

REFERENCES

(1999) Albany Harbours Planning Strategy: Final Guidelines. Water and Rivers Commission, Albany, Western Australia

Aikins S, Kikuchi E (2001) Studies on habitat selection by amphipods using artificial substrates within an estuarine environment. *Hydrobiologia* 457:77-86

Alcoverro T, Duarte CM, Romero J (1995) Annual growth dynamics of *Posidonia-oceanica* - contribution of large-scale versus local factors to seasonality. Marine Ecology Progress Series 120:203-210

Appleford P, Lucas JS, Southgate PC (2003) General principles. In: Lucas JS, Southgate PC (eds) Aquaculture: farming aquatic animals and plants. Blackwell Publishing, Melbourne, p 11-46

Asmus R, Asmus H (1991) Mussel beds: limiting or promoting phytoplankton? Journal of Experimental Marine Biology and Ecology. 148:215-232

Asmus RM, Asmus H, Wille A, Zubillaga GF, Reise K (1994) Complimentary oxygen and nutrient fluxes in seagrass beds and mussel banks?

Barnes R (1987) Chapter 14: Crustaceans. In: Invertebrate Zoology. Saunders College Publishing, Sydney

Barnes RH, R. (1993) An Introduction to Marine Ecology. (2nd ed.) Blackwell Scientific Publications. Melbourne.

Barranguet C (1997) The role of microphytobenthic primary production in a Mediterranean mussel culture area. Estuarine, Coastal and Shelf Science 44:753-765

Barranguet C, Cervetto G, Fontaine MF (1996) Microphytobenthos production in a Mediterranean mussel farm: The influence of incident light. Reports of the Science Academy series III - Life Sciences 319:51-56

Baudinet D, Alliot, E., Berland, B., Grenz, C., Plante-Cuny, M., Plante, R., & Picard, C. (1990) Incidence of mussel culture on biogeochemical fluxes at the sediment-water interface. *Hydrobiologia* 207:187-196

Beasley C, Maguire G (2000) Farming Mussels Aquaculture W.A., p 1-7

Beesley PL, Ross GJB, Glasby CJ (eds) (2000) Polychaetes & Allies: The Southern Synthesis., Vol 4APolychaeta, Myzostomida, Pogonophora, Echiura, Sipunicula. CSIRO Publishing, Melbourne

Bell JD, Westoby M (1986) Abundance of macrofauna in dense seagrass is due to habitat preference, not predation. Oecologia 68:205-209

Bell JD, Westoby M (1987) Effects of an epibenthic alga on abundances of fish and decapods associated with the seagrass *Zostera capricorni*. Australian Journal of Ecology 12:333-337

Beveridge M, Ross L, Kelly L (1994) Aquaculture and biodiversity. *Ambio* 23:497-502

Black KD (2001) Environmental impacts of aquaculture, Vol. Sheffield Academic Press Ltd, Sheffield

Bologna P, Heck KL (1999) Macrofauna associations with seagrass epiphytes - relative importance of trophic and structural characteristics. Journal of Experimental Marine Biology and Ecology 242:21-39

Bostrom C, Bonsdorff E, Kangas P, Norkko A (2002) Long-term changes of a brackish-water eelgrass (*Zostera marina L.*) community indicate effects of coastal eutrophication. Estuarine, Coastal and Shelf Science 55:795-804

Brown LR (2000) Fish farming may soon overtake cattle ranching as a food source. Report No. 9, Earth Policy Institute, Washington DC

Cambridge M, Hocking P (1997) Annual primary production and nutrient dynamics of the seagrasses *Posidonia sinuosa* and *Posidonia australis* in south-western Australia. Aquatic Botany 59:277-295

Cambridge ML, Chiffing AW, Brittan C, Moore L, McComb AJ (1986) The loss of seagrass in Cockburn Sound, Western Australia.II. Possible causes of seagrass decline. Aquatic Botany 24:269-285

Cambridge ML, McComb AJ (1984) The loss of seagrass in Cockburn Sound, Western Australia.I. The time course and magnitude of seagrass decline in relation to industrial development. Aquatic Botany 20:229-243

Cardell MJ, Sarda R, Romero J (1999) Spatial changes in sub-littoral soft bottom polychaete assemblages due to river inputs and sewage discharges. *Acta Oecologica* 20:343-351

Carrasco FD, Carbajal W (1998) The distribution of polychaete feeding guilds in organic enriched sediments of San Vincente Bay, Central Chile. International Review of Hydrobiology 83:233-249

Castell J, Labourg PJ, Escaravage V, Auby I, Garcia ME (1989) Influence of seagrass beds and oyster parks on the abundance and biomass patterns of meio- and macrobenthos in tidal flats. Esturine Coastal and Shelf Science 28:71-85

Ceccherelli G, & Cinelli, F. (1997) Short-term effects of nutrient enrichment of the sediment and interactions between the seagrass *Cymodocea nodosa* and the introduced alga *Caulerpa taxifolia* in a Mediterranean bay. Journal of Experimental Marine Biology and Ecology. 217:165-177

Chamberlain J, T.F. F, P. R, Nickell TD, Davies IM (2001) Impacts of biodeposits from the suspended mussel (*Mytilus edulis L.*) culture on the surrounding surficial sediments. ICES Journal of Marine Science 58:411-416

Clarke K (1993) Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18:117-143

Clarke KR, Warwick RM (1994) Changes in marine communities: an approach to statistical analysis and interpretation. Plymouth Marine Laboratory U.K.

Clesceri L, Greenbrug A, Eaton A (1998) Standard methods for the examination of wastewater. 20th Edition

Connell EL, Walker DI (2001) Nutrient cycling associated with the seagrass Halophila ovalis in the Swan-Canning Estuary based on seasonal variations in biomass and tissue nutrients. Hydrological Processes 15:2401-2409

Connell JH (1978) Diversity in tropical rainforests and coral reefs. Science 199:1302-1310

Connolly R (1997) Differences in composition of small, motile invertebrate assemblages from seagrass and unvegetated habitats in a southern Australian estuary. *Hydrobiologia* 346:137-148

Connolly RM (1995) Effects of removal of seagrass canopy on the assemblages of small, motile invertebrates. Marine Ecology Progress Series 118:129-137

Connolly RM, Butler AJ (1996) The effects of altering seagrass canopy height on small, motile invertebrates of shallow Mediterranean embayments. Marine Ecology 17:637-652

Cruz-Rivera E, Hay ME (2000) The effects of diet mixing on consumer fitness: Macroalgae, epiphytes, and animal matter as food for marine amphipods. Oecologia 123:252-264

D'Adamo (2000) The influence of prevailing winds on water currents in King George Sound, Albany, Western Australia. Personal communication.

D'Adamo N, Simpson C, Mills D, Imberger J, McComb A (1992) The influence of stratification on the ecological response of two Western Australian embayments to nutrient enrichment. Science of the Toal Environment 1992

De Casabianca M, Laugier T, Marinho-Soriano E (1997a) Impact of shellfish farming eutrophication on the benthic macrophyte communities in the Thau lagoon, France. Aquaculture International. 5:301-314

De Casabianca M, Laugier T, Marinho-Soriano E (1997b) Seasonal changes of nutrients in water and sediment in a Mediterranean lagoon with shellfish activity (Thau Lagoon, France). Journal of Marine Science 54:905-916

Delgado O, Grau A, Pou S, Riera F, Massuti C, Zabala M, Ballesteros E (1997) Seagrass regression caused by fish culture in Fornells Bay (Menorca, Western Mediterranean). *Oceanologica Acta* 20:557-563

Delgado O, Ruiz J, Perez M, Romero J, Ballesteros E (1999) Effects of fish farming on seagrass (*Posidonia oceanica*) in a Mediterranean Bay: seagrass decline after organic loading cessation. *Oceanologica Acta* 22:109-117

Dial R, Roughgarden J (1998) Theory of marine communities: the intermediate disturbance hypothesis. Ecology 79:1412-1424

Dimech M, Borg JA, Schembri PJ (2000) Structural changes in a *Posidonia Oceanica* meadow exposed to a pollution gradient from a marine fish-farm in Malta (Central Mediterranean). Biol. Mar. Medit. 7:361-364

Downing J, Anderson M (1985) Estimating the standing biomass of aquatic Macrophytes. Canadian Journal of Fisheries and Aquatic Science 42:1860-1869

Duarte CM (1991) Seagrass depth limits. Aquatic Botany 40:363-377

Edgar G, Robertson A (1992) The influence of seagrass structure on the distribution and abundance of mobile epifauna: pattern and process in a Western Australian Amphibolis bed. Journal of Experimental Marine Biology and Ecology. 160:13-31

Edgar GJ (1990) Population regulation, population dynamics and competition amongst mobile epifauna associated with seagrass. Journal of Experimental Marine Biology and Ecology 144:205-234

Edgar GJ (2000) Australian Marine Life, Vol. Read-New Holland, Sydney

F.A.O. (2002) The State of World Fisheries and Aquaculture: Part 1, World review of fisheries and aquaculture, Food and Agriculture Organisation of the United Nations: Fisheries Department, Rome

Fabiano M, Danovaro R, Olivari E, Misic C (1994) Decomposition of faecal matter and somatic tissue of *Mytilus galloprovincialis*: changes in organic matter composition and microbial succession. Marine Biology:375-384

Ferrell D, Bell J (1991) Differences among assemblages of fish associated with *Zostera capricorni* and bare sand over a large spatial scale. Marine Ecology Progress Series 72:15-24

Filskov P (1990) Oxygen excretion. In: Phillips RC, McRoy CP (eds) Seagrass research methods: monographs on oceanographic methodology., Vol 9. United Nations Educational, Paris

Findlay R, & Watling, L. (1997) Prediction of benthic impact for salmon net-pens based on the balance of bethic oxygen supply and demand. Marine Ecology Progress Series 155:147-157

Findlay RH, Watling L (1995) Environmental impacts of salmon net-pen culture on marine communities in Maine: A case study. Estuaries 18:145-179

Fitzpactrick J, & Kirkman, F. (1995) Effects of prolonged shading stress on growth and survival of seagrass *Posidonia australis* in Jervis Bay, New South Whales, Australia. Marine Ecology Progress Series 127:279-289

Folke C, Kautsky N (1989) The role of ecosystems for sustainable development of aquaculture. Ambio 18:234-243

Frankovich T, & Fourquurean (1997) Seagrass epiphyte loads along a nutrient availability gradient, Florida Bay, USA. Marine Ecology Progress Series. 159:37-50

Gambi MC, Conti G, Bremec CS (1998) Polychaete distribution , diversity and seasonality related to seagrass cover in shallow soft bottoms of the Tyrrhenian Sea (Italy). *Scientia Marina* 62:1-17

Garcia E, Duarte CM, Middelburg JJ (2002) Carbon and nutrient deposition in a Mediterranean seagrass (*Posidonia oceanica*) meadow. Limnology and Oceanography. 47:23-32

Gibbs MM, Pickmere SE, Woods PH, Payne GW, James MR, Hickman RW, Illingworth J (1992) Nutrient and chlorophyll a variability at six stations associated with farming in Pelorus Sound, 1984-85. New Zealand Journal of Marine and Freshwater Research 26:197-211

Gilbert F, Souchu, P., Bianchi, M., Bonin, P. (1997) Influence of shellfish farming activities on nitrification, nitrate reduction to ammonium and denitrification at the water-sediment interface of the Thau lagoon. Marine Ecology Progress Series 151:143-153

Gordon D, Chase S, Grey K, Simpson C (1994) Changes to the structure and productivity of a *Posidonia sinuosa* seagrass meadow in Princess Royal Harbour, Western Australia, during and after imposed shading. Dept. of Environmental Protection: W.A.

Grant J, Hatcher A, Scott D, Pocklington P, Shafer C, Winters G (1995) A multidisciplinary approach to evaluating impacts of shellfish aquaculture an benthic communities. Estuaries 18:124-144

Grenz C, Hermin M, Baudinet D, Daumas R (1990) In situ biochemical and bacterial variation of sediments enriched with mussel deposits. *Hydrobiologia* 207:153-160

Hall M, Bell S (1988) Response of small motile epifauna to complexity of epiphytic algae on seagrass blades. Journal of Marine Research 46:613-630

Hall M, Bell SS (1993) Meiofauna on the seagrass *Thalassia testudinum* - Population characteristics of Harpacticoid copepods and associations with algal epiphytes. Marine Biology 116:137-146

Hall P, Anderson L, Holby O, Kollberg S, Samuelsson M (1990) Chemical fluxes and mass balance in a marine fish cage farm. I. Carbon. Marine Ecology Progress Series 61:61-73

Hannan JC, Williams RJ (1998) Recruitment of juvenile marine fishes to seagrass habitat in a temperate Australian estuary. Estuaries 21:29-51

Hargrave B, Phillips G, Doucette L, White M, Milligan T, Wildish D, Cranston R (1997) Assessing benthic impacts of organic enrichment from marine aquaculture. Water, Air and Soil Science.:641-650

Hatcher A, Grant J, Schofield B (1994) Effects of suspended mussel culture (*Mytilus spp.*) on sediment nutrient dynamics in a coastal bay. Marine Ecology Progress Series 115:219-235

Hauxwell J, Cebrian J, Valiela I (2003) Eelgrass *Zostera marina* loss in temperate estuaries: relationship to land-derived nitrogen loads and effects of light limitation imposed by algae. Marine Ecology Progress Series 247:59-73

Hemminga M (1998) The roots/rhizome system of seagrasses: and asset and a burden? Journal of Sea Research 39:183-196

Hemminga M, Harrison P, van Lent P (1991) The balance of nutrient losses and gains in seagrass meadows. Marine Ecology Progress Series. 71:85-96

Hemminga MA, Duarte CM (2000) Seagrass Ecology, Vol. Cambridge University Press, Cambridge

Henderson A, Gamito S, Karakassis I, Pederson P, Smaal AC (2001) Use of hydrodynamics and benthic models for managing environmental impacts of marine aquaculture. Journal of Applied Ichthyology 17:163-172

Henderson AR, D. (1995) Use of macrobenthic infaunal communities in the monitoring and control of the impact of marine cage fish farming. Aquaculture Research 26:659-678

Hill BW, T. (1993) Why are some prawns found in seagrass? An experimental study of Brown (*Penaeus esculentus*) and grooved (*P. semisulcatus*) Tiger Prawns. Australian Journal of Marine and Freshwater Research. 44:221-227

Holmer MK, E. (1992) Impact of marine fish cage farming on the metabolism and sulphate reduction of the underlying sediments. Marine Ecology Progress Series 80:191-201

Hovel KA, Fonseca MS, Myer DL, Kenworthy WJ, Whitfield PE (2002) Effects of seagrass landscape structure, structural complexity and hydrodynamic regime on macrofaunal densities in North Carolina seagrass beds. Marine Ecology Progress Series 243:11-24

Hutchings PA, Wells FE, Walker DI, Kendricks GA (1991) Seagrass, sediment and infauna - a comparison of *Posidonia australis*, *Posidonia sinuosa* and *Amphibolis antarctica* in Princess Royal Harbour, south-western Australia. II. Distribution, composition and abundance of macrofauna. Proceedings of the Third International Marine Biological Workshop: The Marine Flora and Fauna of Albany, Western Australia., Western Australian Museum, Perth, p 611-633

Iwama GK (1991) Interactions between aquaculture and the environment. Critical Reviews in Environmental Control. 21:177-216

Jernakoff P, Nielsen J (1997a) Plant-animal associations in two species of seagrasses in Western Australia. Aquatic Botany 60:359-376

Jernakoff P, Nielsen J (1997b) The relative importance of amphipod and gastropod grazers in *Posidonia sinuosa* meadows. Aquatic Botany 56:183-202
Jernakoff P, Phillips BF, Fitzpatrick JJ (1993) The diet of post-parulis western rock lobster, *Panulirus cygnus George*, at Seven Mile Beach, Western Australia. Australian Journal of Marine & Freshwater Research.
Jespersen A-M, Christoffersen K (1987) Measurements of chlorophyll-a from phytoplankton using ethanol as extraction solvent. Archive of *Hydrobiologie* 14:445-454
Jones DS, Morgan GJ (2002) A field guide to crustaceans of Australian waters, Vol. Western Australian Museum / Reed New Holland, Sydney
Josefsen SB, Schluter L (1994) The influence of an intertidal mussel bed (*Mytilus edulis L.*) on nutrient fluxes in the Kerteminde Fjord, Denmark; a flume study.
Karakassis I, Hatziyanni E (2000) Benthic disturbance due to fish farming analysed under different levels of taxonomic resolution. Marine Ecology Progress Series 203:247-253
Kaspar H, Gillespie P, Boyer I, MacKenzie A (1985) Effects of mussel aquaculture on the nitrogen cycle and the benthic communities in Kenepura Sound, Marlborough Sounds, New Zealand. Marine Biology 85:127-136
Kautsky N, Evans S (1987) Role of biodeposition by *Mytilus edulis* in the circulation of matter and nutrients in a Baltic coastal ecosystem. Marine Ecology Progress Series 38:201-212
Kirkman H (1997) Seagrasses of Australia. State of the Environment Australia. Technical Paper Series(Estuaries and the Sea)
Kirkman H, Humphries P, Manning R (1991) The epibenthic fauna of seagrass beds and bare sand in Princess Royal Harbour and King George Sound, Albany south-western Australia. Proceedings of the Third Marine Biological Workshop.:553-557
Kirkman H, Kuo J (1990) Pattern and Processes in southern Western Australian seagrasses. Aquatic Botany 37:367-382
Kirkman H, Reid (1979) A study of the role of the seagrass *Posidonia Australis* in the carbon budget of an estuary. Aquatic Botany 7:173-183
Kroncke I (1996) Impact of biodeposition on macrofaunal communities in intertidal sand-flats. Marine Ecology 17:159-174
Kuo J, Phillips R, Walker DI, Kirkman H (1996) Seagrass Biology. Proceeding of an International Workshop.
La Rosa T, Mirto S, Favaloro E, Savona B, Sara G, Danovaro R, Mazzola A (2000) Impact on the water column biogeochemistry of a Mediterranean mussel and fish farm. Water Research 36:713-721
Lapointe BE, Tomasko DA, Matzie WR (1994) Eutrophication and trophic state classification of seagrass communities in the Florida Keys. Bulletin of Marine Science 54:696-717
Lee SY (1997) Potential trophic importance of the faecal material of the mangrove crab *Sesarma messa*. Marine Ecology Progress Series 159:275-284
Lee SY, Fong CW, Wu RSS (2001) The effects of seagrass (*Zostera japonica*) canopy structure on associated fauna: a study using artificial seagrass units and sampling of natural beds. Journal of Experimental Marine Biology and Ecology 259:23-50
Lopez N, Duarte, C., Vallespinos, F., Romero, J., & Alcoverro, T. (1998) The effects of nutrient additions on the bacterial activity in seagrass (*Posidonia oceanica*) sediments. Journal of Experimental Marine Biology and Ecology. 224:155-166
MacIntyre H, Geider R, Miller D (1996) Microphytobenthos: The ecological role of the "Secret Garden" of unvegetated, shallow-water marine habitats. I. Distributions, abundances and primary production. Estuaries 19:186-201
Maguire G (2002) State Of The Fisheries: Mussel Farming Status, Department of Fisheries,, Perth, Western Australia
Marba N, Cebrian J, Enriquez S, Duarte CM (1996) Growth patterns of western Mediterranean seagrasses: Species-specific responses to seasonal forcing. Marine Ecology Progress Series 133:203-215
Martinova MV (1993) Nitrogen and phosphor compounds in the bottom sediments: mechanisms of accumulation, transformation and release. *Hydrobiologia* 252:1993

Martin-Smith KM (1993) Abundance of mobile epifauna - The role of habitat complexity and predation by fishes. Journal of Experimental Marine Biology and Ecology 174:243-260
Masini RJ, Manning CR (1997) The photosynthetic responses to irradiance and temperature of four meadow-forming seagrasses. Aquatic Botany 58:21-36
Mattsson J, Linden O (1983) Benthic macrofauna succession under mussels *Mytilus Edulis L.* (*bivalvia*), cultures on hanging long-lines. *Sarsia* 68:97-102
Mazouni N, Gaertner, J., Deslous-Paoli, J., Landrein, S., d'Oedenberg, G. (1996) Nutrient and oxygen exchanges at the water-sediment interface in a shellfish farming lagoon (Thua, France). Journal of Experimental Marine Biology and Ecology. 205:91-113
McGuinness KA (2002) Of rowing boats, ocean liners and tests of the ANOVA homogeneity of varience assumption. Austral Ecology 27:681-688
McMahon K, & Walker, D.I. (1998) Fate of seasonal, terrestrial nutrient inputs to a shallow seagrass dominated embayment. Estuarine, Coastal and Shelf Science. 46:15-25
Meese R, Tomich P (1992) Dots on the rocks: a comparison of present cover estimation methods. Journal of Experimental Marine Biology and Ecology. 165:59-73
Mendez S, Pergent G, Pergent-Martoni C (1997) Impact of fish farming facilities on coastal ecosystems. In: Ozhan E (ed) Third International Conference on the Mediterranean Coastal Environment: MEDCOAST 97., Qawra, Malta.
Meteorology (1999) Wind roses using available data between 1990 and 1999 for Albany airport. Albany office of the Bureau of Meteorology
Midlen A, Redding T (1998) Aquaculture Series 2: Environmental management for aquaculture, Vol. Kluwer Academic Publishers, Dordrecht
Miller MW, Sluka RD (1999) Patterns of seagrass and sediment nutrient distribution suggest anthropogenic enrichment in Laamu Atoll, Republicc of Maldives. Marine Pollution Belletin 38:1152-1156
Mirto S, La Rosa T, Danovaro R, Mazzola A (2000) Microbial and meiofaunal response to intensive mussel-farm biodeposition in coastal sediment of the Western Mediterranean. Marine Pollution Bulletin 40:244-252
Nakaoka M, Toyohara T, Matsumasa M (2001) Seasonal and between-substrate variation in motile epifaunal community in a multispecific seagrass bed of Otsuchi Bay, Japan. Marine Ecology 22:379-395
Naylor RL, Goldburg RJ, Primavera JH, Kautsky N, Beveridge MCM, Clay J, Folke C, Lubchenco J, Mooney H, Troell M (2000) Effects of aquaculture on world fish supplies. Nature 405:1017-1024
Ogilvie SC, H. RA, Schiel DR (2000) Phytoplankton biomass associated with mussel farms in Beatrix Bay, New Zealand. Aquaculture 181:71-80
Olsson P, Graneli E (1992) Structuring of a post-spring phytoplankton community by manipulation of trophic interactions. Journal of Experimental Marine Biology and Ecology 158:249-266
Orth RJ, Heck J, K.L., Montfrans JV (1984) Fauna communities in seagrass beds: a review of the influence of plant structure and prey characteristics on predator-prey relationships. Estuaries 7:1984
Parker JD, Duffy JE, Orth RJ (2001) Plant species diversity and composition: experimental effects on marine epifaunal assemblages. Marine Ecology Progress Series 224:55-67
Pearson T, Rosenburg R (1978) Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology, Annual Review. 16:229-311
Pergent G, Mendez S, Pergent-Martini C, Pasqualini V (1999) Preliminary data on the impact of fish farming facilities on *Posidonia oceanica* meadows in the Mediterranean. *Oceanologica Acta.* 22:95-107
Peterson BJ, Heck Jr KL (1999) The potential for suspension feeding bivalves to increase seagrass productivity. Journal of Experimental Marine Biology & Ecology. 240:37-52
Peterson BJ, Heck Jr KL (2001a) An experiment test of the mechanism by which suspension feeding bivalves elevate seagrass productivity. Marine Ecology Progress Series 218:115-125

Peterson BJ, Heck Jr KL (2001b) Positive interactions between suspension-feeding bivalves and seagrass - a facultative mutualism. Marine Ecology Progress Series 213:143-155
Pillay T (1992) Aquaculture and the Environment.
Prins TC, Smaal AC (1994) The role of the blue mussel *Mytilus edulis* in the cycling of nutrient in the Oosterschelde estuary (The Netherlands). *Hydrobiologia* 282/283:413-429
Prins TC, Smaal AC, Pouwer AJ, Dankers N (1996) Filtration and resuspension of particulate matter and phytoplankton on an intertidal mussel bed in the Oosterschelde estuary (SW Netherlands). Marine Ecology Progress Series 142:121-134
Radziejewska T (1986) On the role of *Mytilus edulis* aggregations in enhancing meiofauna communities off the southern Baltic coast. Ophelia 4
Ragnarsson SA, Raffaelli D (1999) Effects of the mussel *Mytilus edulis L.* on the invertebrate fauna of sediments. Journal of Experimental Marine Biology and Ecology 241:31-43
Reusch TBH, Chapman A, Groger J (1994) Blue mussels *Mytilus edulis* do not interfere with eelgrass *Zostera marina*, but fertilise shoot growth through biodepostion. Marine Ecology Progress Series 108:265-282
Reusch TBH, Williams SL (1998) Variable responses of native eelgrass *Zostera marina* to a non-indigenous bivalve *Musculista senhousia. Oecologia* 113:428-441
Ritz DA, Lewis ME, Shen M (1989) Response to organic enrichment of infaunal macrobenthic communities under salmonoid sea-cages. Marine Biology 103:211-214
Robertson AI, Mann KH (1980) The role of isopods and amphipods in the initial fragmentation of eelgrass detritus in Nova Scotia, Canada. Marine Biology 59:63-69
Rossi F (2002) Dynamics of the polychaetes *Desdemona ornata* and *Perinereis cultrifera* and their resources of food in a Mediterranean Mudflat. Marine Ecology 23:285-296
Ruitton S, Francour P, Boudouresque CF (2000) Relationships between algae, benthic herbivorous invertebrates and fishes in rocky sublittoral communities of a temperate sea (Mediterranean). Estuarine, Coastal and Shelf Science 50:217-230
Ruiz JM, Perez M, Romero J (2001) Effects of fish farm loadings on seagrass (*Posidonia oceanica*) distribution, growth and photosynthesis. Marine Pollution Bulletin 42:749-760
Ruiz JM, Romero J (2001) Effects of insitu. experimental shading on the Mediterranean seagrass *Posidonia oceanica*. Marine Ecology Progress Series 215:107-120
Ruiz JM, Romero J (2003) Effects of disturbances caused by coastal constructions on the spatial structure, growth dynamics and photosynthesis of the seagrass *Posidonia oceanica*. Marine Pollution Bulletin 46:1523-1533
Sala E, Boudouresque CF (1997) The role of fishes in the organization of a Mediterranean sublittoral community .1. Algal communities. Journal of Experimental Marine Biology and Ecology 212:24-44
Samuelson GM (2001) Polychaetes as indicators of environmental disturbance on subarctic tidal flats, Iqaluit, Baffin Island, Nunavut Territory. Marine Pollution Bulletin 42:733-741
Sanchez-Jerez P, Barbera-Cebrian C, Ramos-Espla AA (2000) Influence of the structure of *Posidonia oceanica* meadows modified by bottom trawling on crustacean assemblages: comparison of amphipods and decapods. *Scientia Marina* 64:319-326
Schneider FI, Mann KH (1991) Species specific relationships of invertebrates to vegetation in a seagrass bed. II. experiments on the importance of macrophyte shape, epiphyte cover, and predation. Journal of Experimental Marine Biology and Ecology 145:119-139
Sergeev VN, Clarke SM, Sheperd SA (1988) Motile macroepifauna of the seagrasses, *Amphibolis* and *Posidonia*, and unvegetated sandy substrata in Holdfast Bay, South Australia. Transactions of the Royal Society of South Australia 112:97-108
Short F (1986) Effects of sediment nutrients on seagrasses: literature review and mesocosm experiment. Aquatic Botany 27:41-57
Short F, Wyllie-Echeverria S (1996) Natural and human induced disturbances of seagrasses. Environmental Conservation 23:17-27
Short FT, Coles RG (2001) Global Seagrass Research Methods, Vol. Elsevier, London

Silberstein K, Chiffings AW, McComb AJ (1986) The loss of seagrass in Cockburn Sound, Western Australia.111.The effect of epiphytes on productivity of *Posidonia Australis*. Aquatic Botany 24:355-371

Silvert W (1992) Assessing environmental impacts of finfish aquaculture in marine waters. Aquaculture 107:67-79

Smaal AC, Zurburg W (1996) The uptake and release of suspended and dissolved material by oysters and mussels in Marennes-Oleron Bay. Aquatic Living Resources 10:23-30

Snelgrove PVR, Austen MC, Boucher G, Heip C, Hutchings PA, King GM, Koike I, Lambshead JD, Smith CR (2000a) Linking biodiversity above and below the marine sediment-water interface. BioScience 50:1076-1088

Snelgrove PVR, Austen MC, Boucher G, Heip C, Hutchings PA, King GM, Koike IP, Lambshead JD, Smith CR (2000b) Linking Biodiversity above and below the sediment-water interface. BioScience 50:1076

Snelgrove PVR, Butman CA (1994) Animal-sediment relationships revisited: causes versus effects. Oceanography and Marine Biology: An annual review 32:111-177

Sogard SM, Powell GVN, Holmquist JG (1987) Epibenthic fish communities on Florida Bay banks: relations with physical parameters and seagrass cover. Marine Ecology Progress Series 40:25-39

Stenton-Dozey J, Probyn T, Busby A (2001) Impact of mussel (*Mytilus galloprovincialis*) raft-culture on benthic macrofauna, insitu. oxygen uptake, and nutrient fluxes in saldanha Bay, South Africa. Canadian Journal of Fisheries and Aquatic Science 58:1021-1031

Sundback K, Viveka, E., Wilhelm, G., & Karin, P. (1991) Influence of sublittoral microphytobenthos on the oxygen and nutrient flux between sediment and water: a laboratory continuous-flow study. Marine Ecology Progress Series. 74:263-279

Swift D (1993) Aquaculture Training Manual. Blackwell Scientific Publications, Victoria

Tenore KR, Boyer LF, Cal RM, Corral J, Garcia-Fernandez C, Gonzalez N, Gonzalez-Gurriaran E, Hanson RB, Iglesias J, Krom M, Lopoz-Jamar E, McClain J, Pamatmat MM, Perez A, Rhodes DC, de Santiago G, Tietjen J, Westrich J, Windom HL (1982) Coastal up-welling in the Rias Bajas, NW Spain: Contrasting the benthic regimes of the Rias de Arosa and de Muros. Journal of Marine Research 40:701-772

Touchette BW (2000) Review of nitrogen and phosphorus metabolism in seagrass. Journal of Experimental Marine Biology and Ecology 250:133-167

Turner S, Grant, J., Pridmore, R., Hewitt, J., Wilkinson, M., Hume, T., & Morrisey, D. (1997) Bed-load and water-column transport and colonization processes by post-settlement benthic macrofauna: Does infauna density matter? Journal of Experimental Marine Biology and Ecology. 216:51-75

Udy J, & Dennison, W. (1997a) Physiological responses of seagrass used to identify anthropogenic nutrient inputs. Marine and Freshwater Research. 48:605-614

Udy J, & Dennison. (1997b) Growth and physiological responses of three seagrass species to elevated sediment nutrients in Moreton Bay, Australia. Journal of Experimental Marine Biology and Ecology. 217:253-277

Underwood A, J. (1997) Experiments in Ecology, Their logical Design and Interpretation Using Analysis of Variance., Vol. Cambridge University Press, Cambridge

Valentine JF, Heck J, K.L. (1993) Mussels in seagrass meadows: their influence on macroinvertebrate abundance and secondary production in the northern Gulf of Mexico. Marine Ecology Progress Series 96:63-74

Valiela I (1995) Marine Ecological Processes, Vol. Springer, New York

Walker D, McComb A (1992) Seagrass degradation in Australian Waters. Marine Pollution Bulletin 25:191-195

Warwick RM (1988) The level of taxonomic discrimination required to detect pollution effects on marine benthic communities. Marine Pollution Bulletin 19:259-268

Warwick RM, Pearson TH, Rushwahyuni. (1987) Detection of pollution effects on marine macrobenthos: further evaluation of the species abundance biomass method. Marine biology 95:193-200

Webster PJ (1998) Effect of shoot density on the infaunal macro-invertebrate community within a *Zostera marina* Seagrass bed. Estuarine Coastal and Shelf Science 47:351-357

Weston D (1990) Quantitative examination of macrobenthic community changes along an organic enrichment gradient. Marine Ecology Progress Series. 61:233-244

Wilson WH (1981) Sediment mediated interactions in a densely populated infaunal assemblage: the effects of the polychaete *Abarenicola pacifica.* Journal of Marine Research 39:221-241

Wood N, Lavery P (2000) Monitoring seagrass ecosystem health - The role of perception in defining health and indicators. Ecosystem Health 6:134-148

Wu R, Lam, K., MacKay, D., Lau, T., Yam, V. (1994) Impact of marine fish farming an water quality and bottom sediment; a case study in the sub-tropical environment. Marine Environmental Research. 38:115-145

Wu RSS (1995) The environmental impact of marine fish culture: towards a sustainable future. Marine Pollution Bulletin 31:159-166

Yokoyama H (2002) Impact of fish and pearl farming on the benthic environments in Gokasho Bay: Evaluation from seasonal fluctuations of the macrobenthos. Fisheries Science 68:258-268

APPENDIX

Plate 1. Misery Beach, 300m southwest of the mussel-line aquaculture

Plate 2. Cyroidiedae 1 mm

Plate 3. Amaryllidae 1 mm

Plate 4. Iphimediidae 1 mm

Plate 5. Taxon S (unidentified) 1 mm

Plate 6. Ischyroceridae 1 mm

Plate 7. Melitidae 1 mm

Plate 8. Lysianassidae 1 mm

Platye 9. Isaidae 1 mm

Plate 10. Ochlesidae 1 mm

Plate 11. Dexaminidae 1 mm

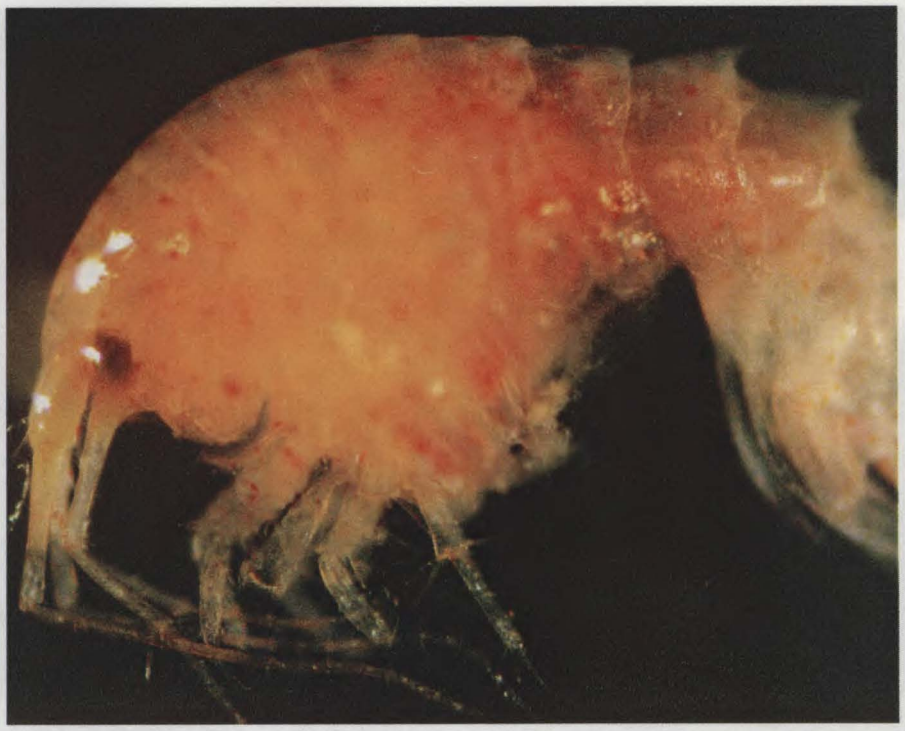


Plate 12. Dexaminidae 1 mm

Plate 13. Diogenidae 1 mm

Plate 14. Gnathidae (male) 1 mm

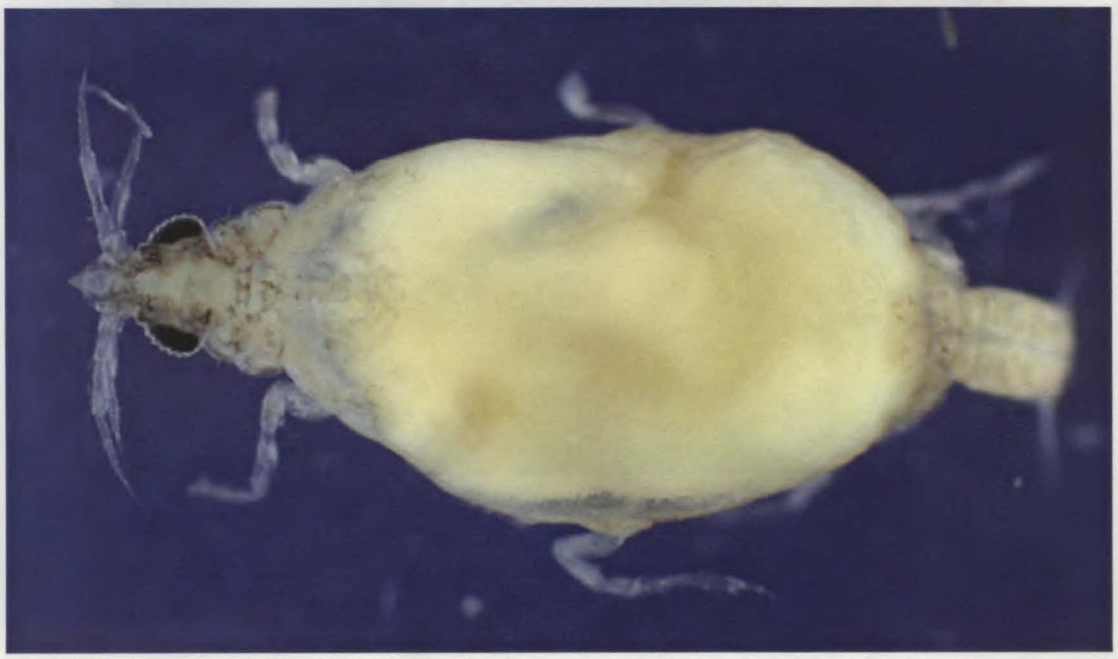


Plate 15. Gnathidae (female) 1 mm

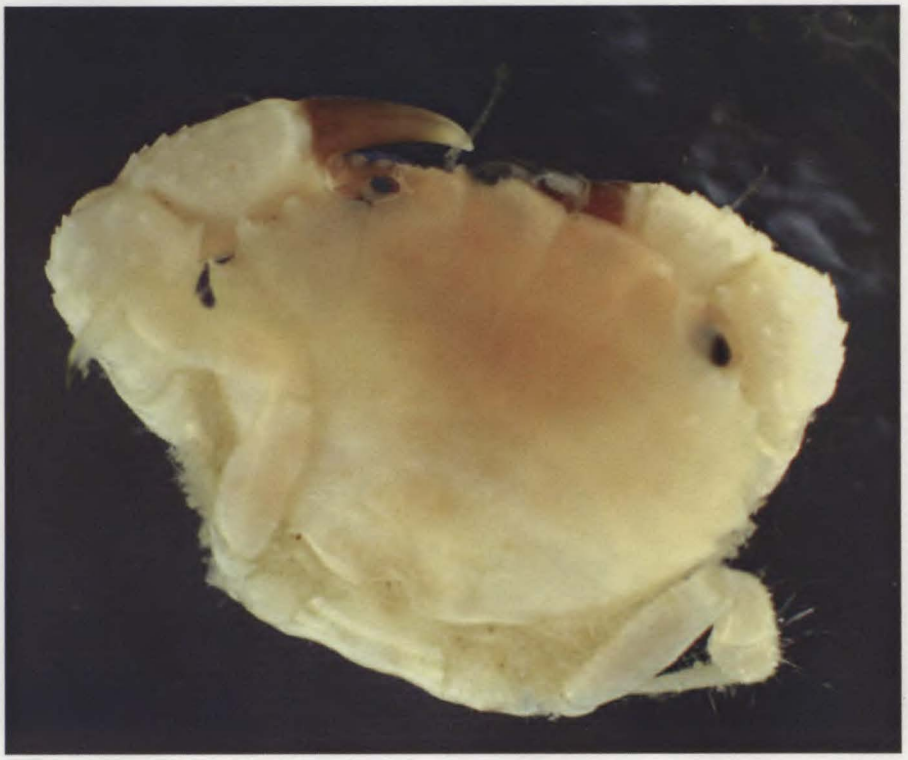


Plate 16. Majidae 1 mm

Plate 17. Majidae 1 mm

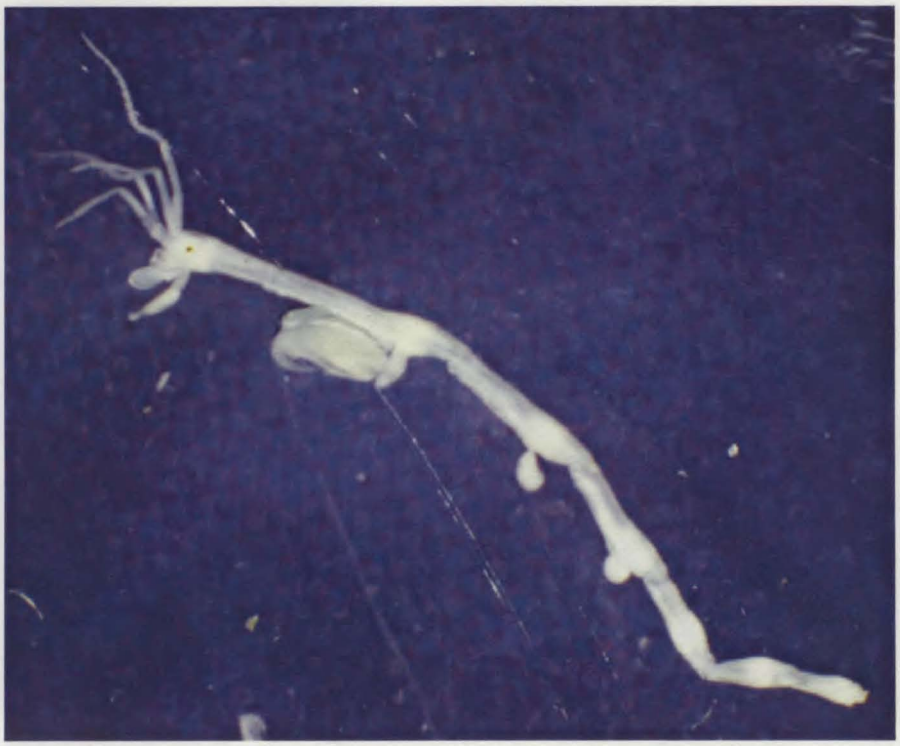


Plate 18. Caprellidae 1 mm

Plate 19. Phtisicidae 1 mm

Plate 20. Phtisicidae 1 mm

Plate 21. Protellidae 1 mm

Plate 22. Ostracod sp. 1 mm

Plate 23. Unidentified taxon 1 mm

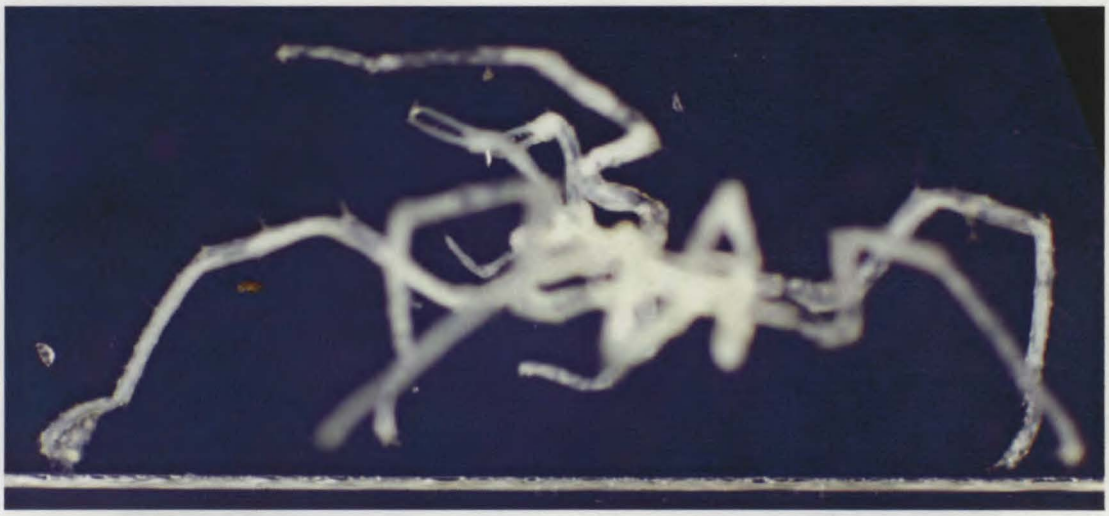


Plate 24. Picnogonidae 1 mm

Plate 25. Stenetridae 1 mm

Plate 26. Anomuran sp. 1 mm

Plate 27. Phliantidae 0.1 mm

Plate 28. Tanaid sp. 0.1 mm

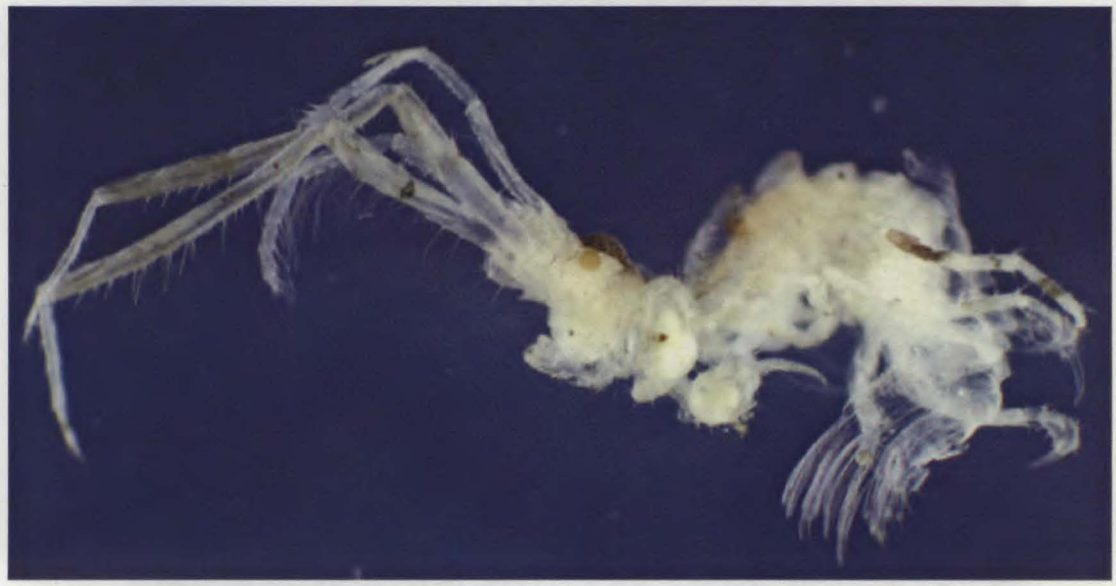


Plate 29. Podoceridae 1 mm

Plate 30. Taxon X (unidentified) 1 mm

Plate 31. Arcturidae 1 mm

Figure 1. Leucothoidae

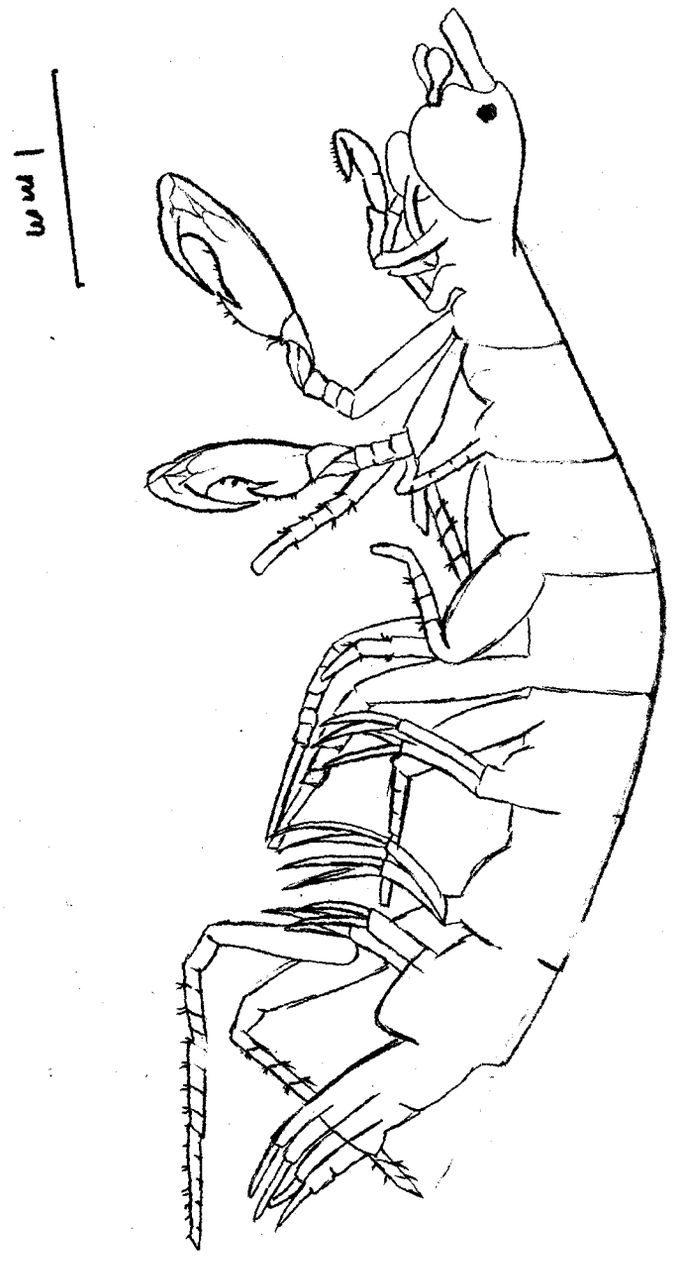


Figure L. Dexaminidae

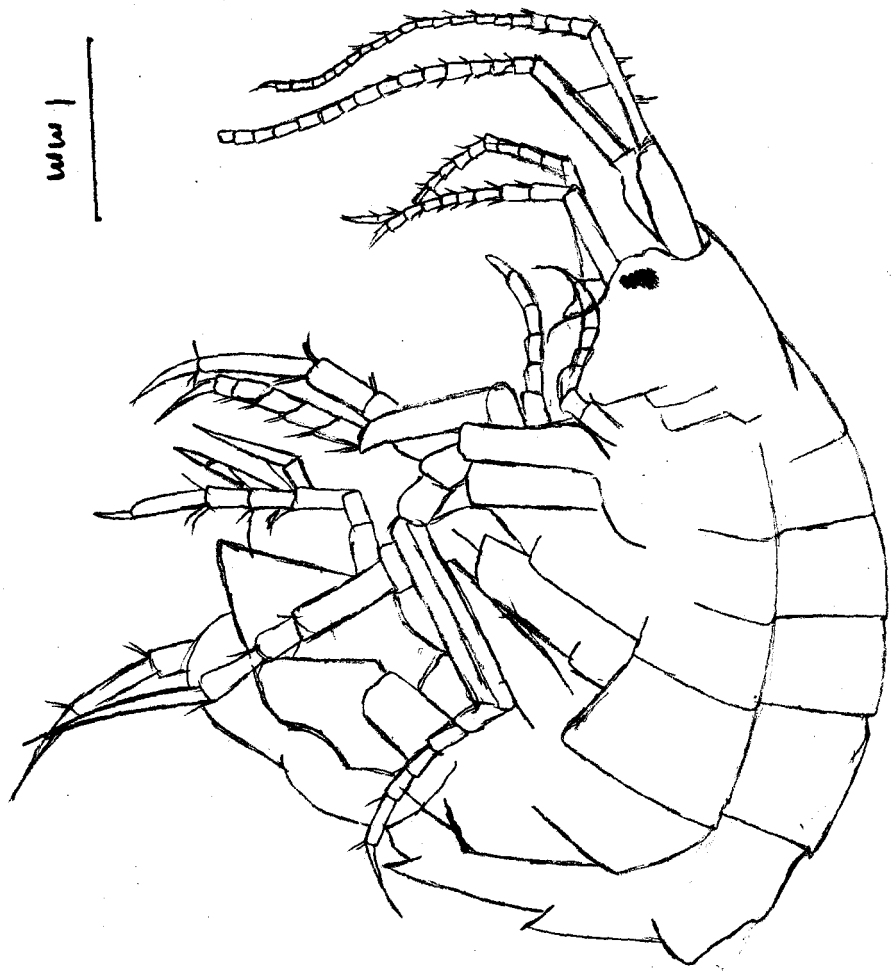


Figure 3. Continue

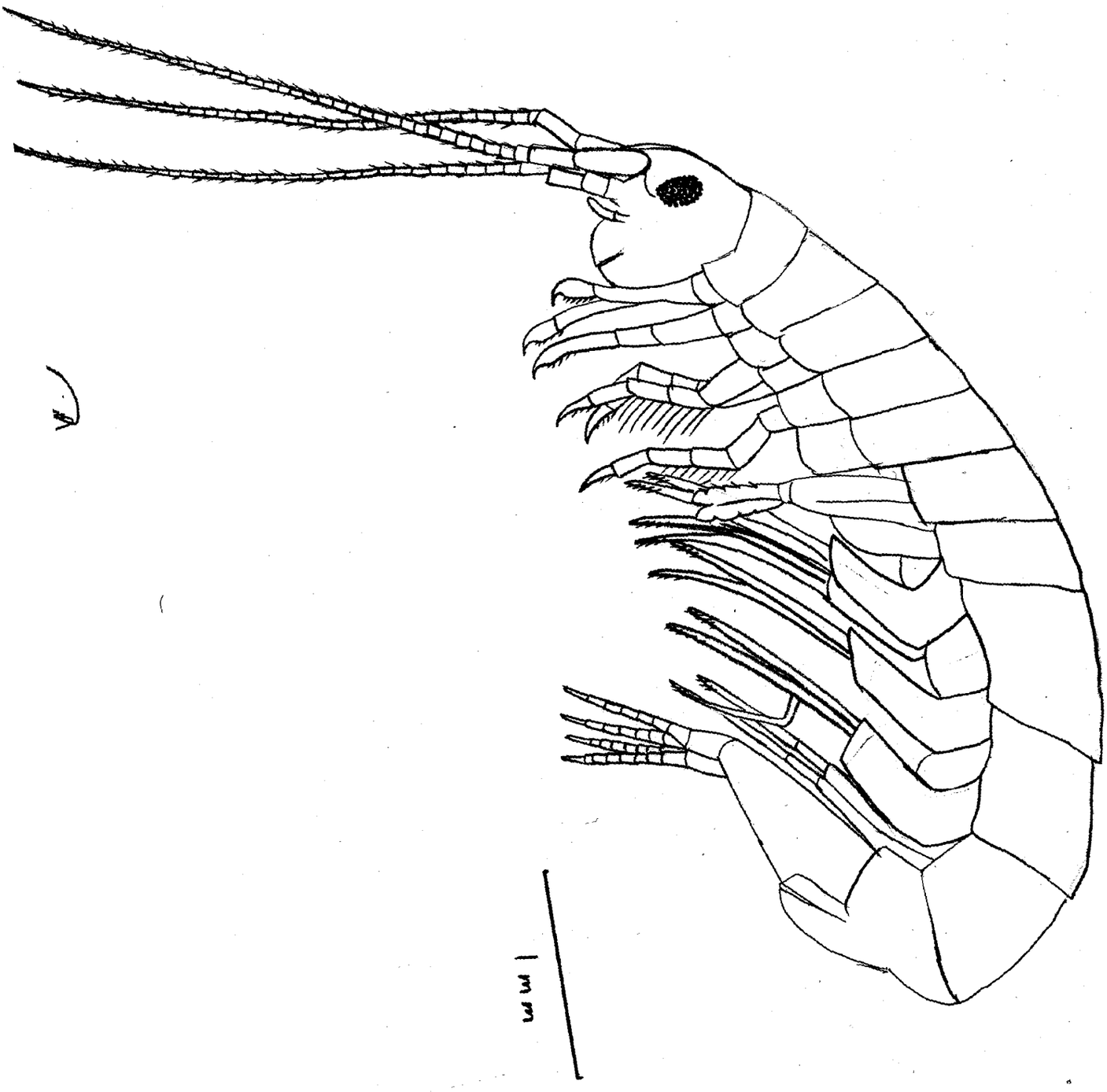


Figure 4. (Unidentified

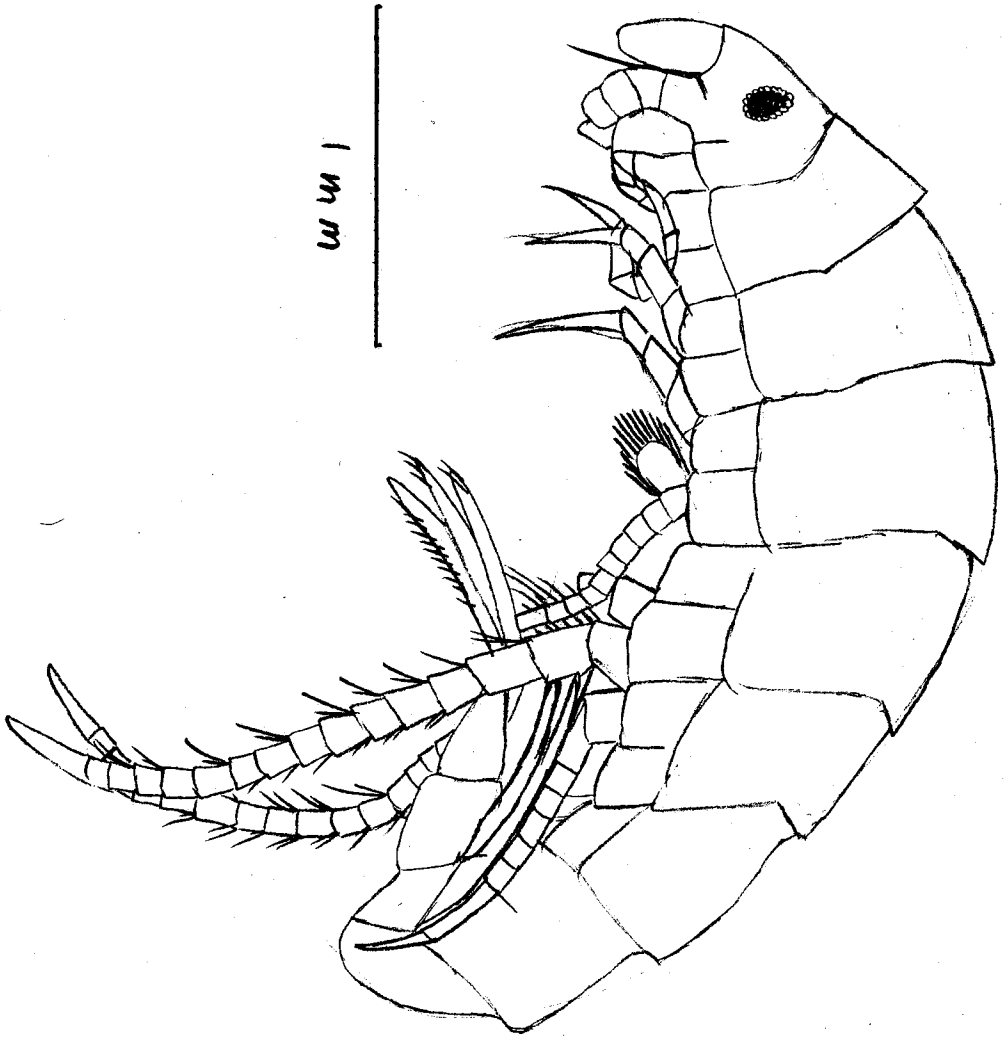


Figure 5. Cyprolimnadae

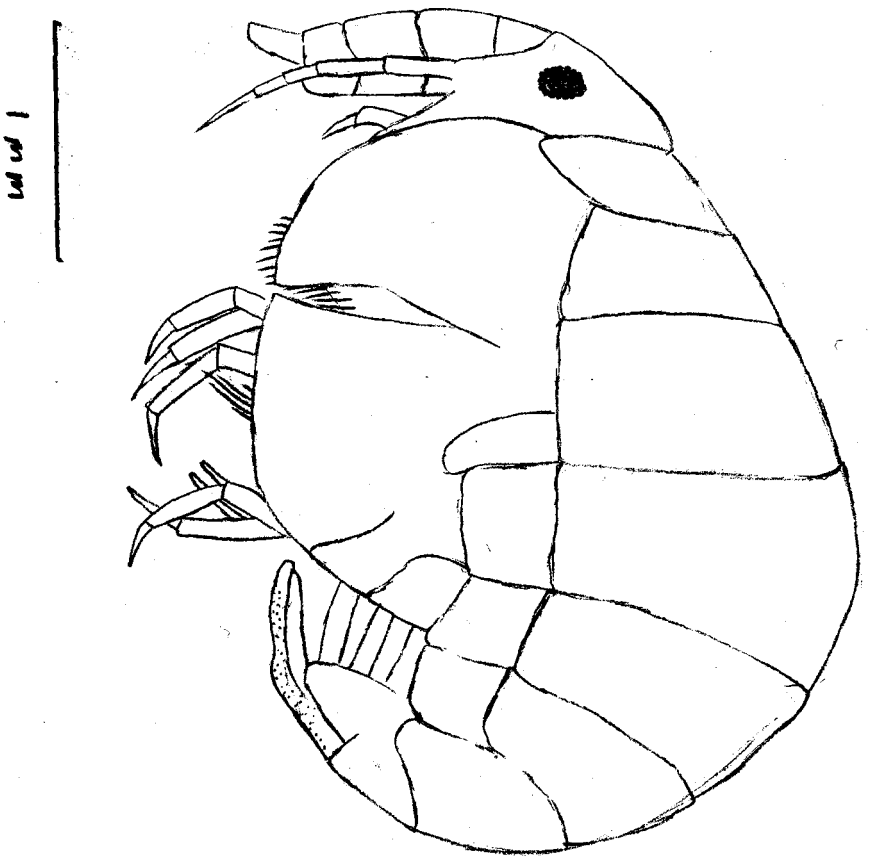


Figure 6. Hemimidae

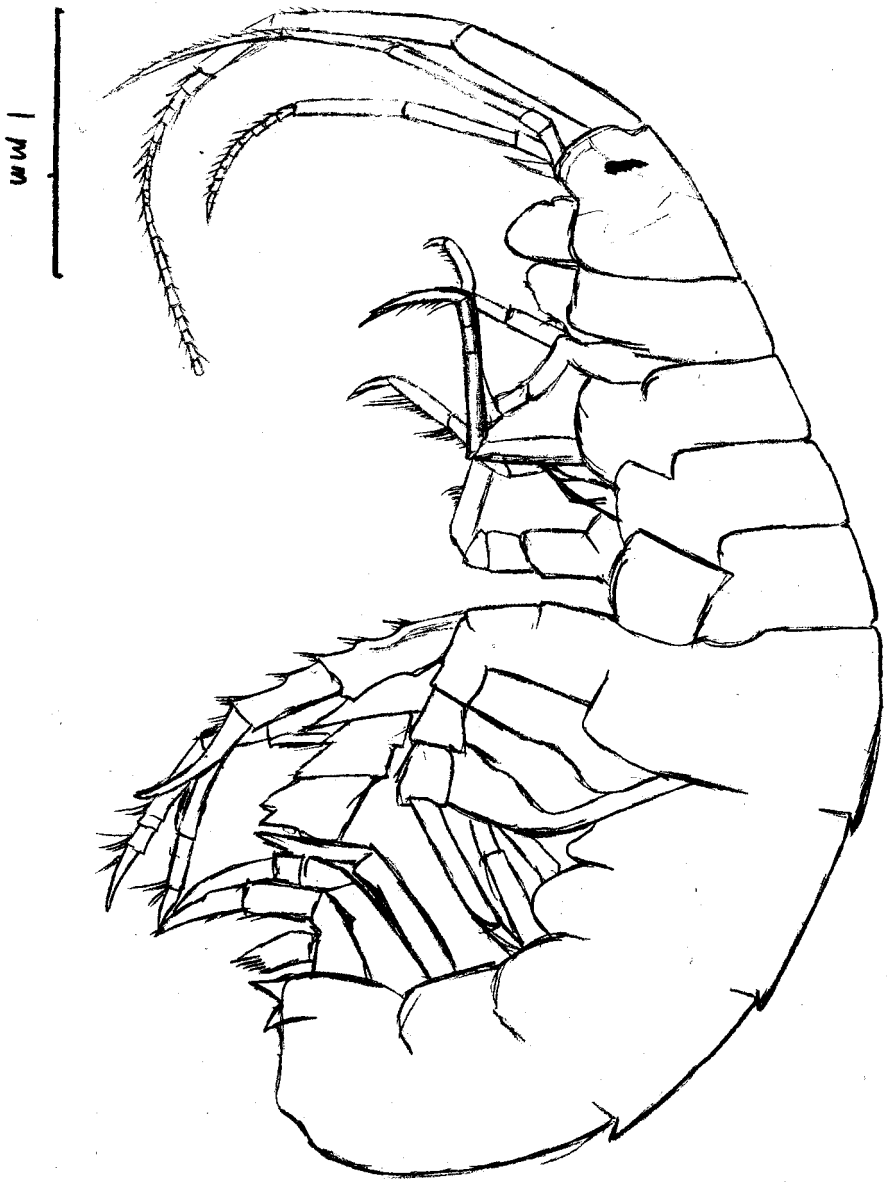
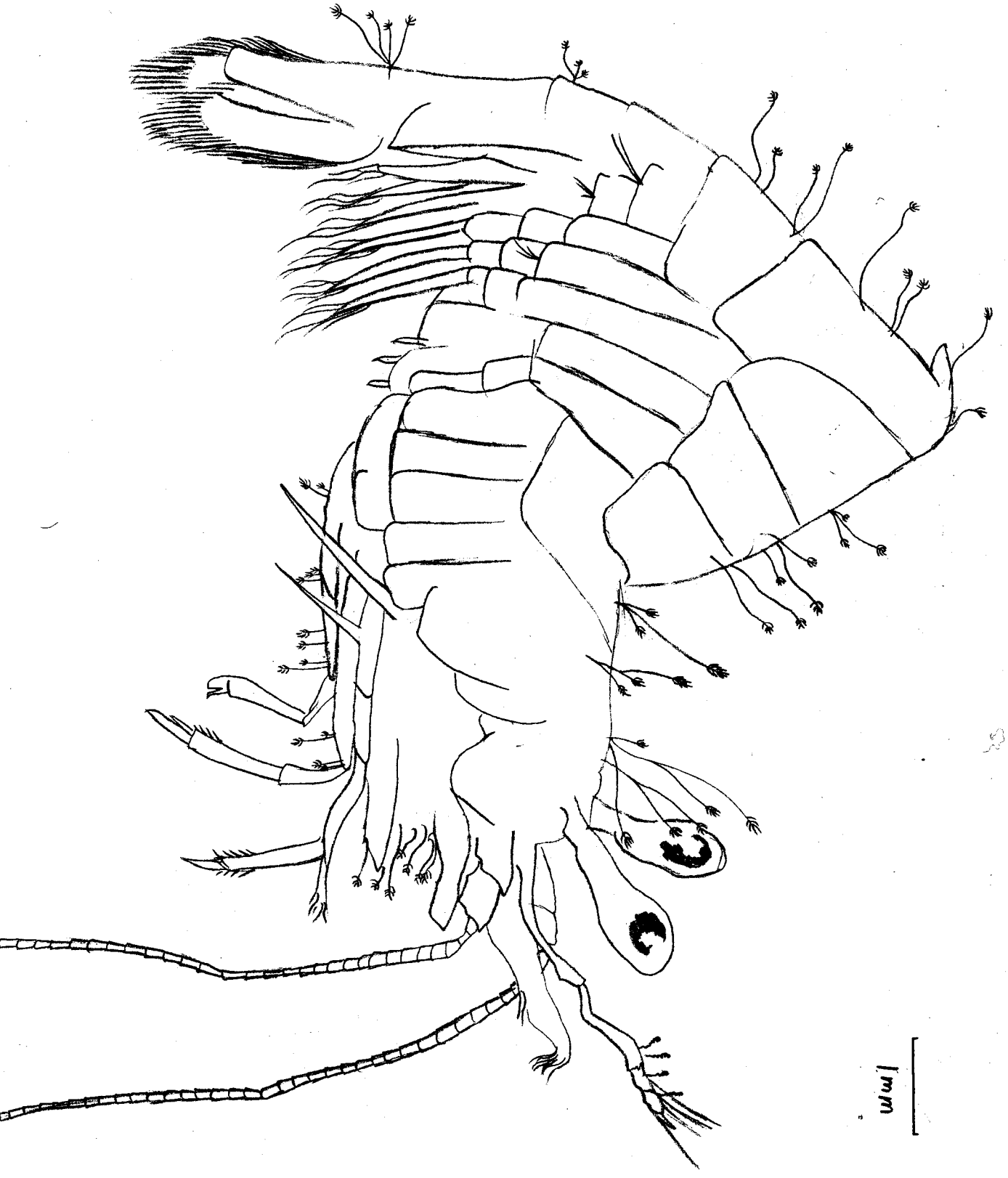


Figure 7. Hippolytidae



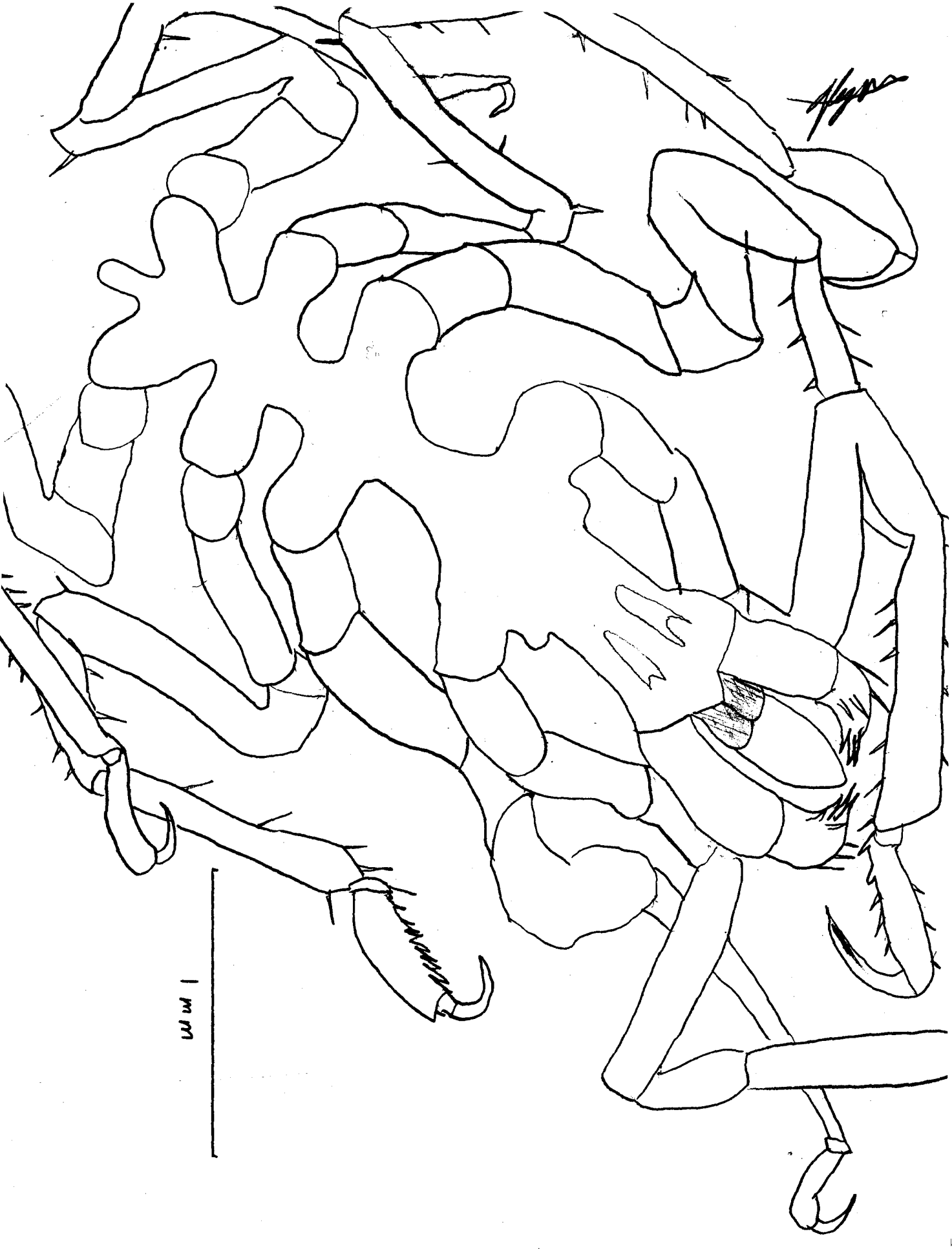
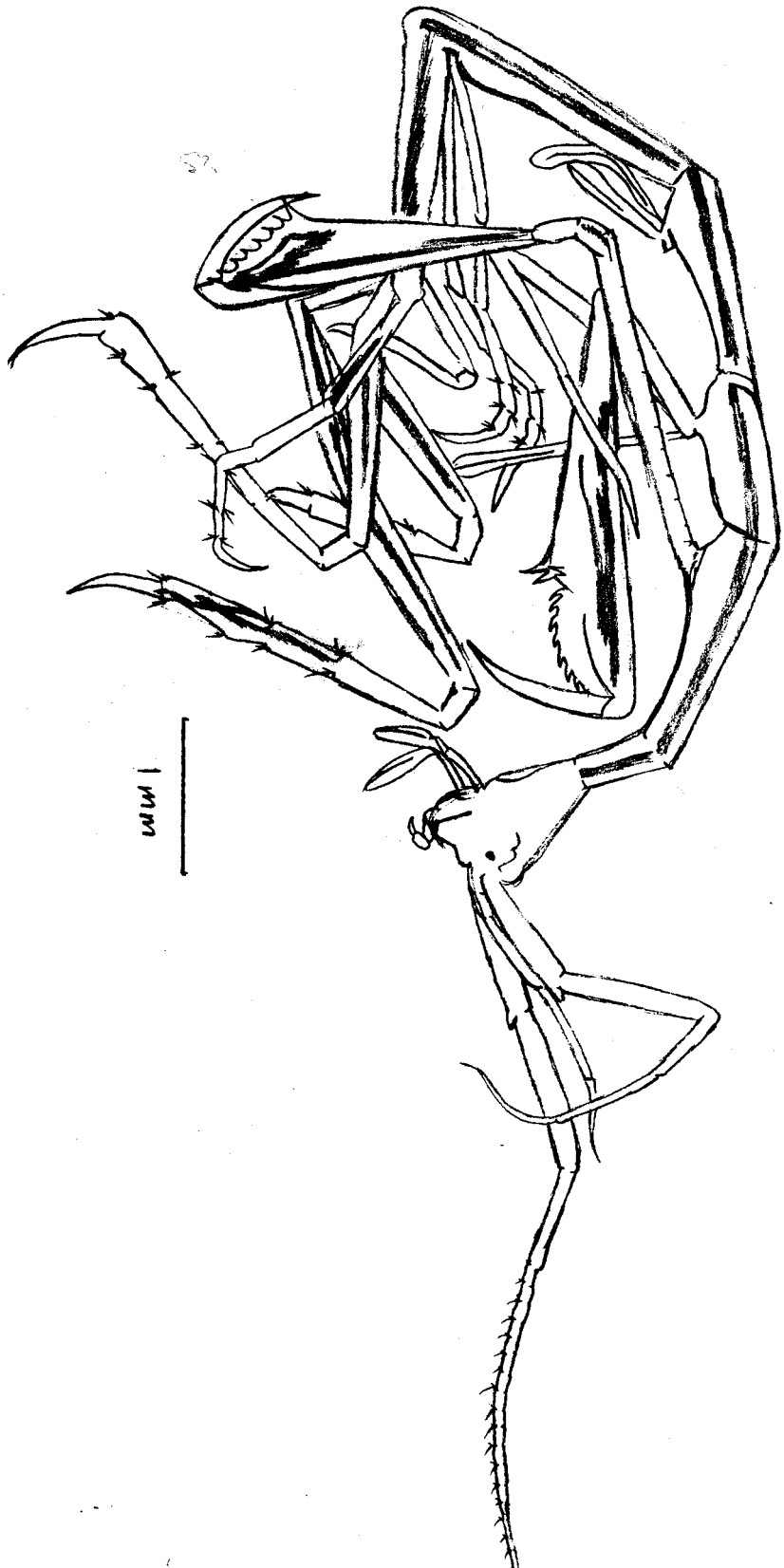


Figure 8. Picnogonidae

Figure 9. Phtisicidae



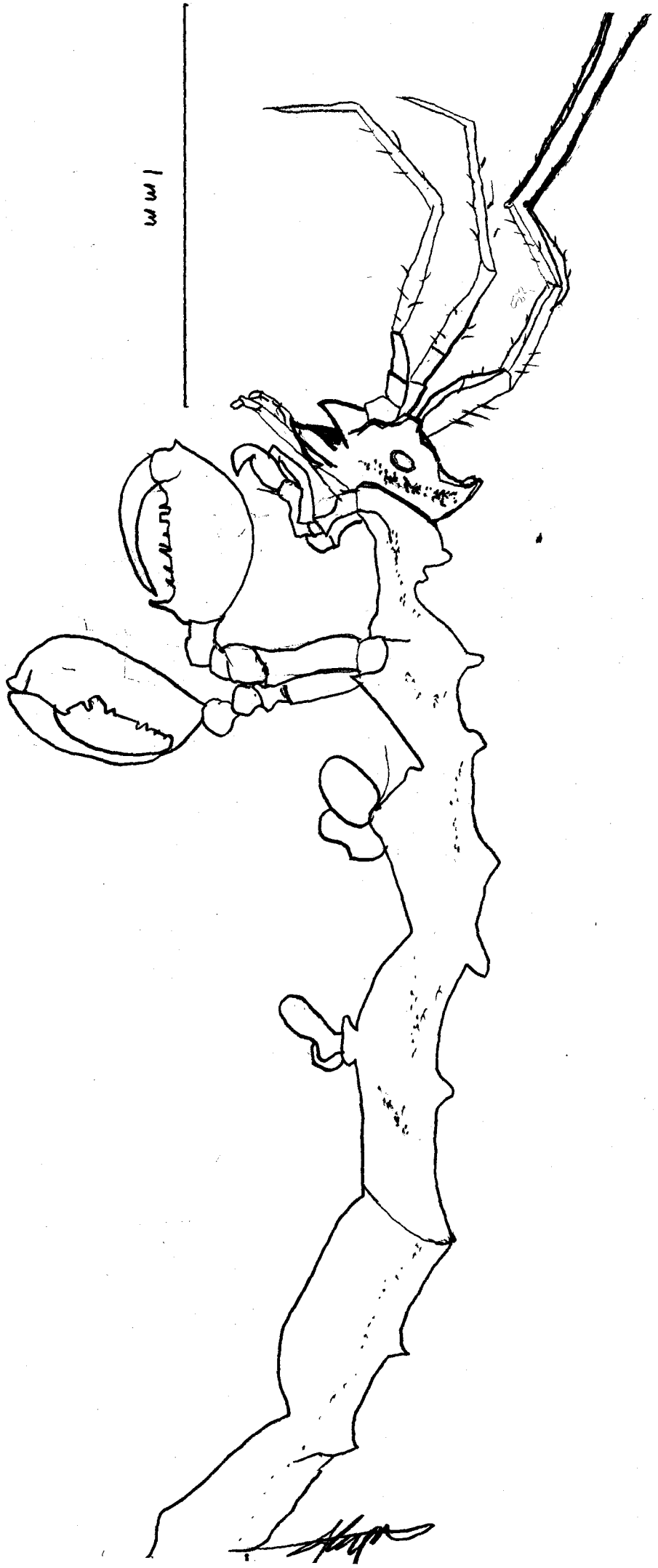
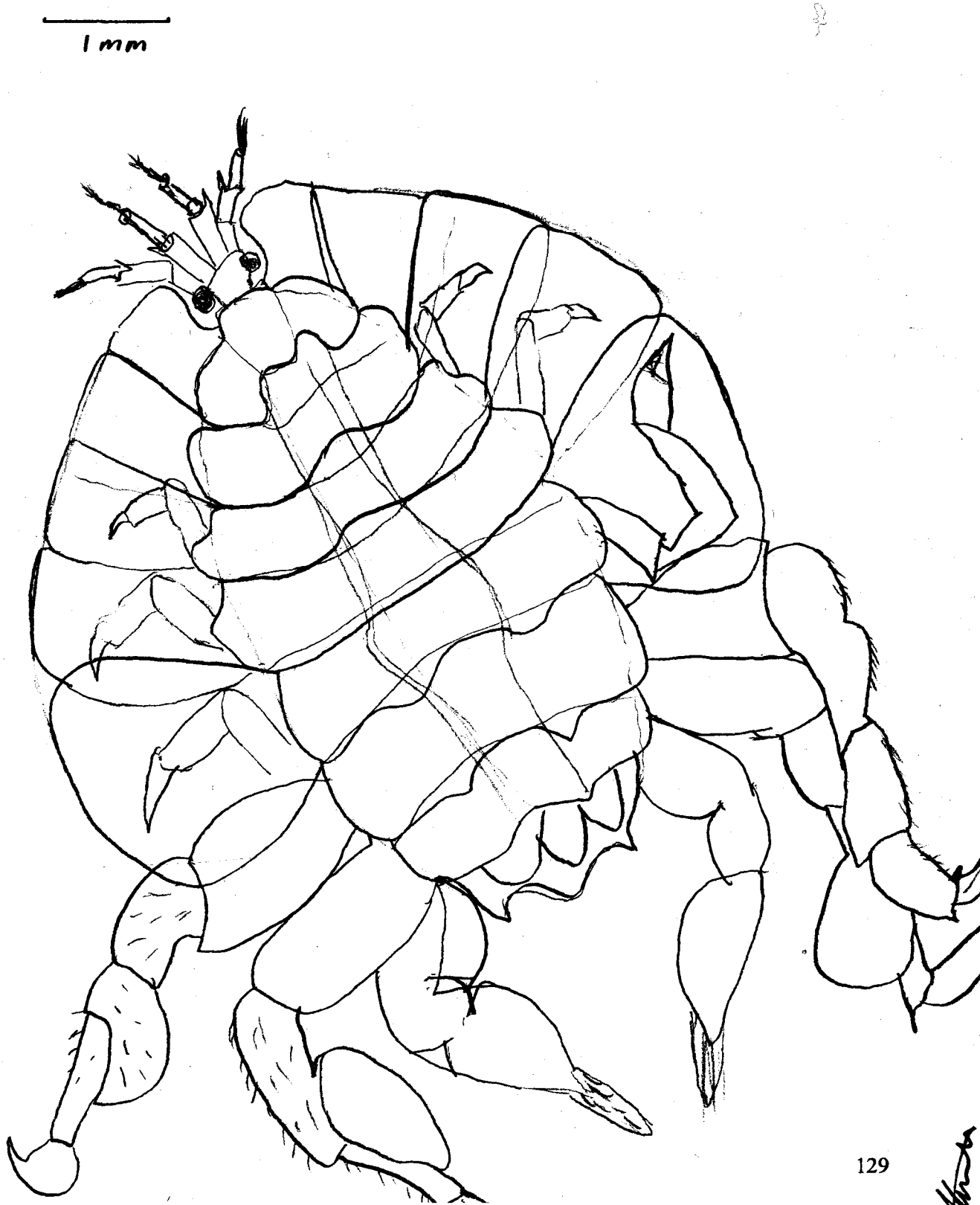


Figure 10.
Protellidae

Figure 11. Thliantiqae



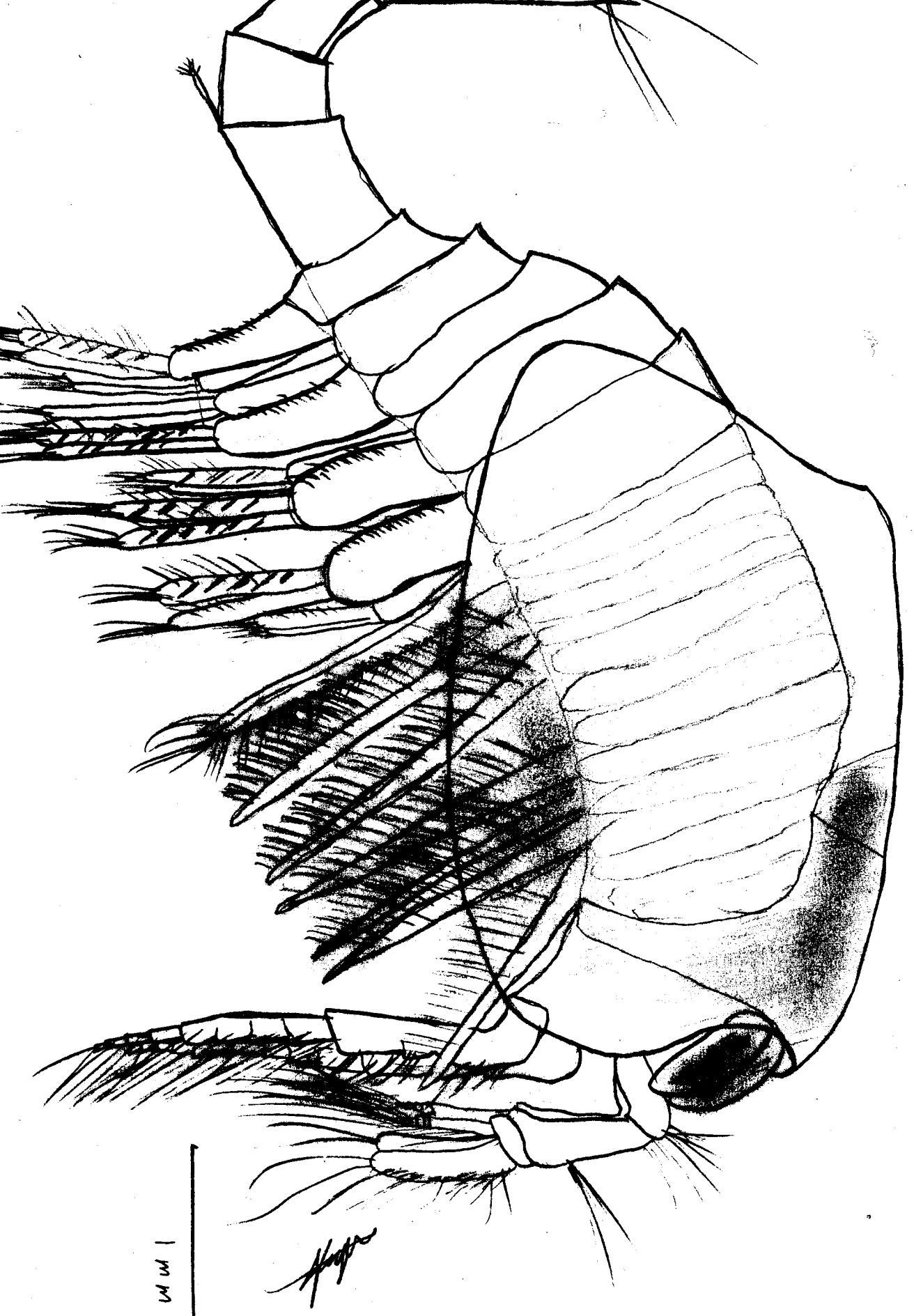
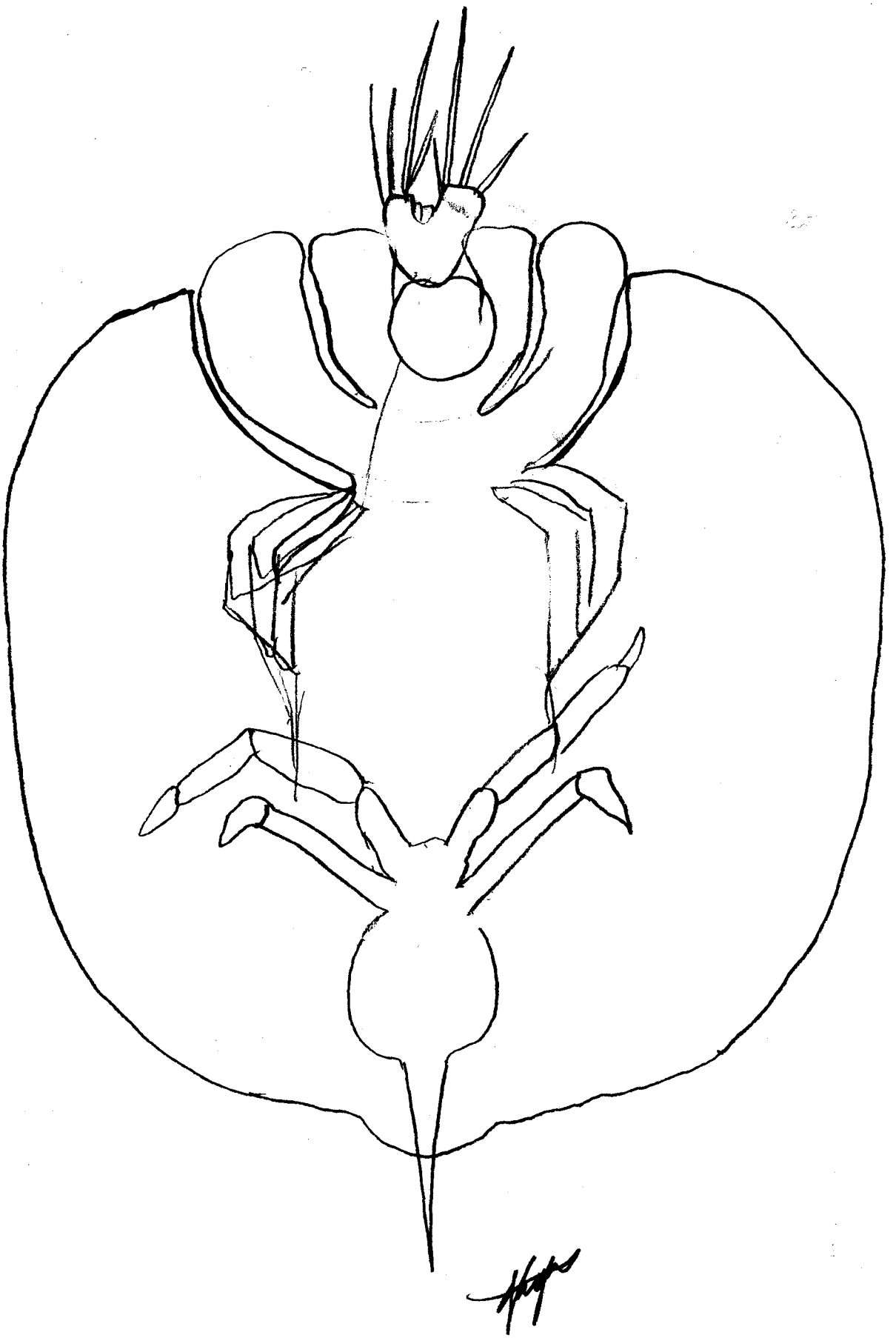


Figure 12. Nebalidae

Figure 13. (unidentified)



1 mm

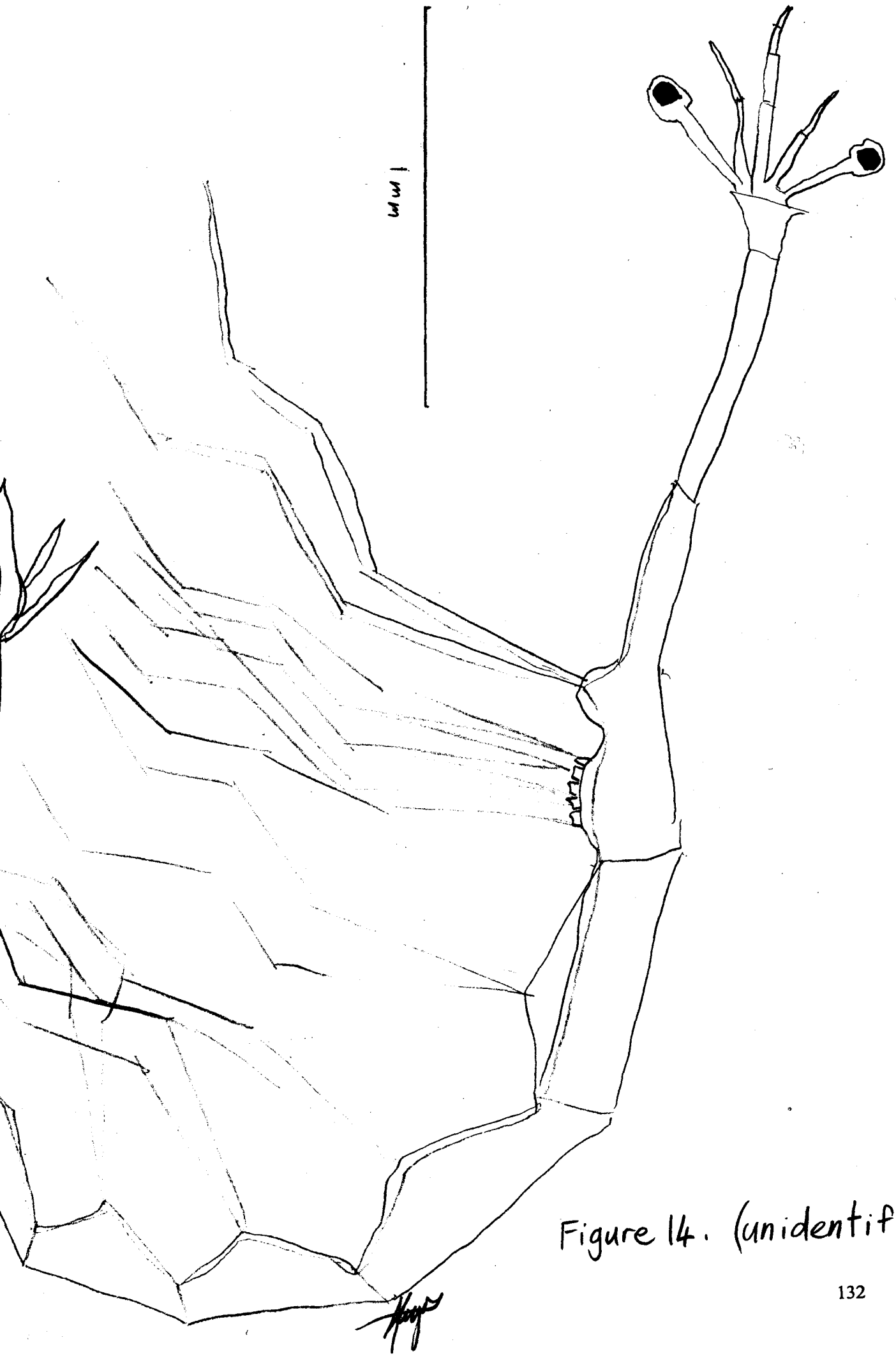


Figure 14. (unidentifie

Figure 15. Gnathiidae (male)

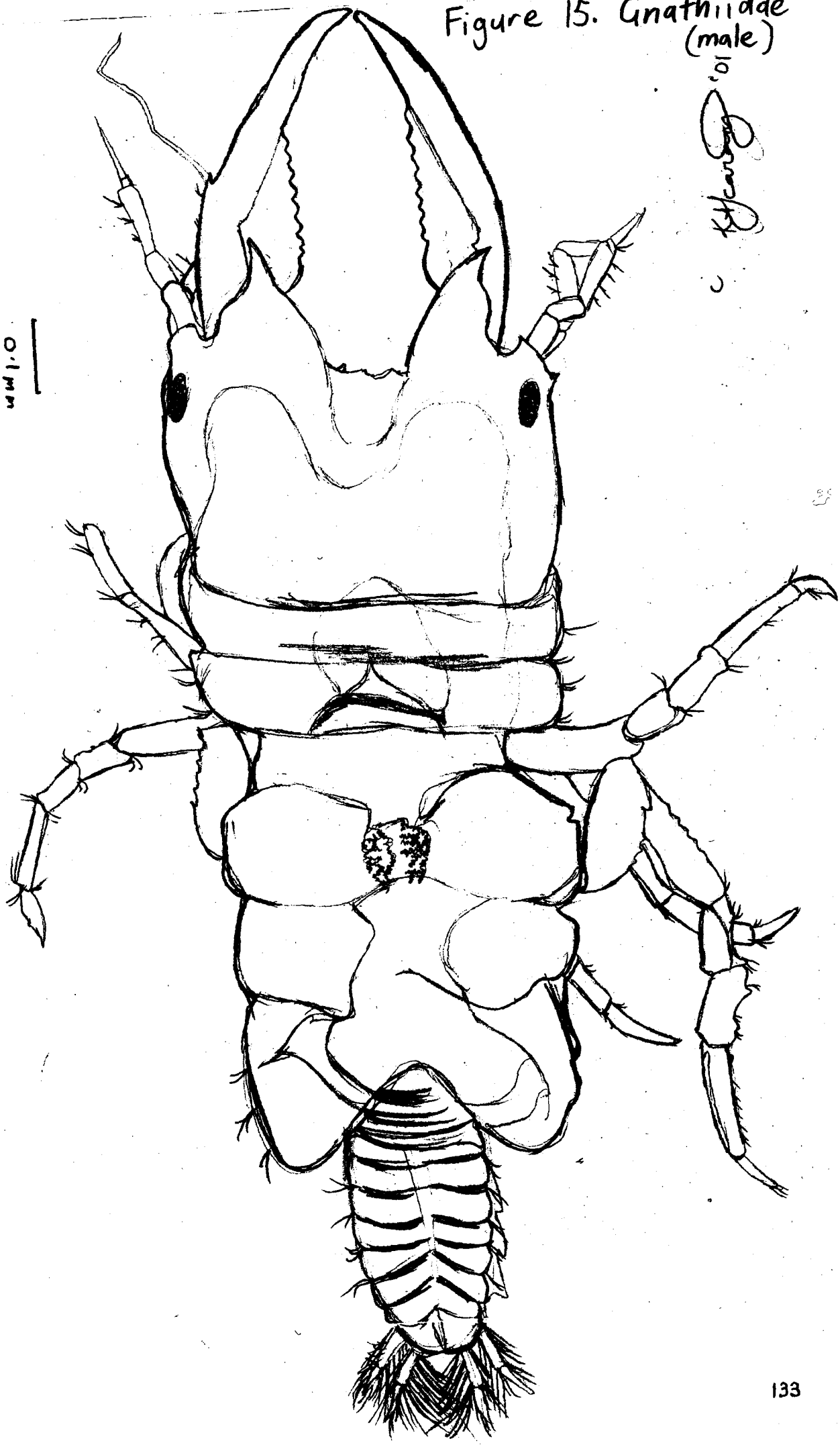
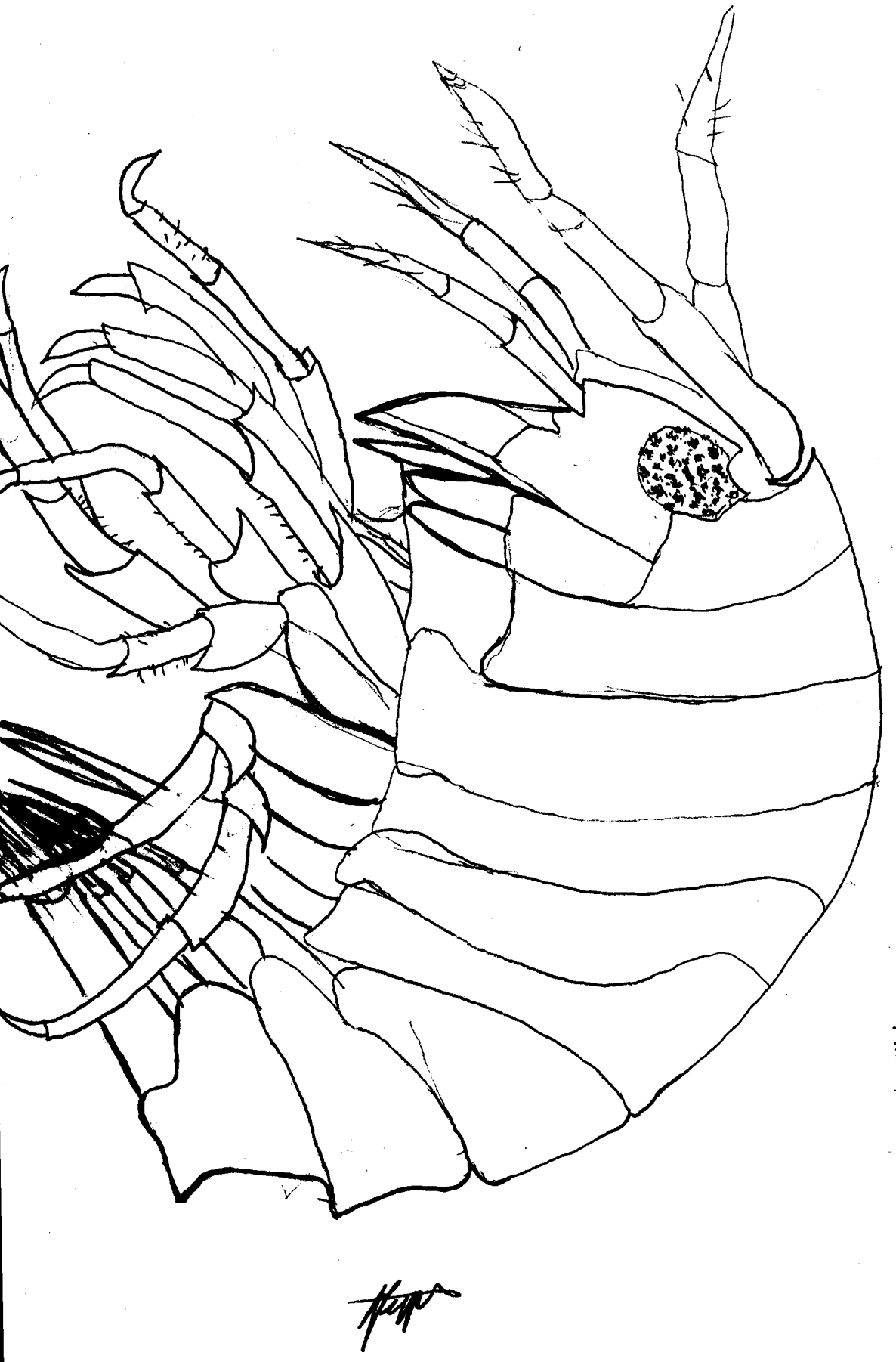


Figure 16. Ochlesidae



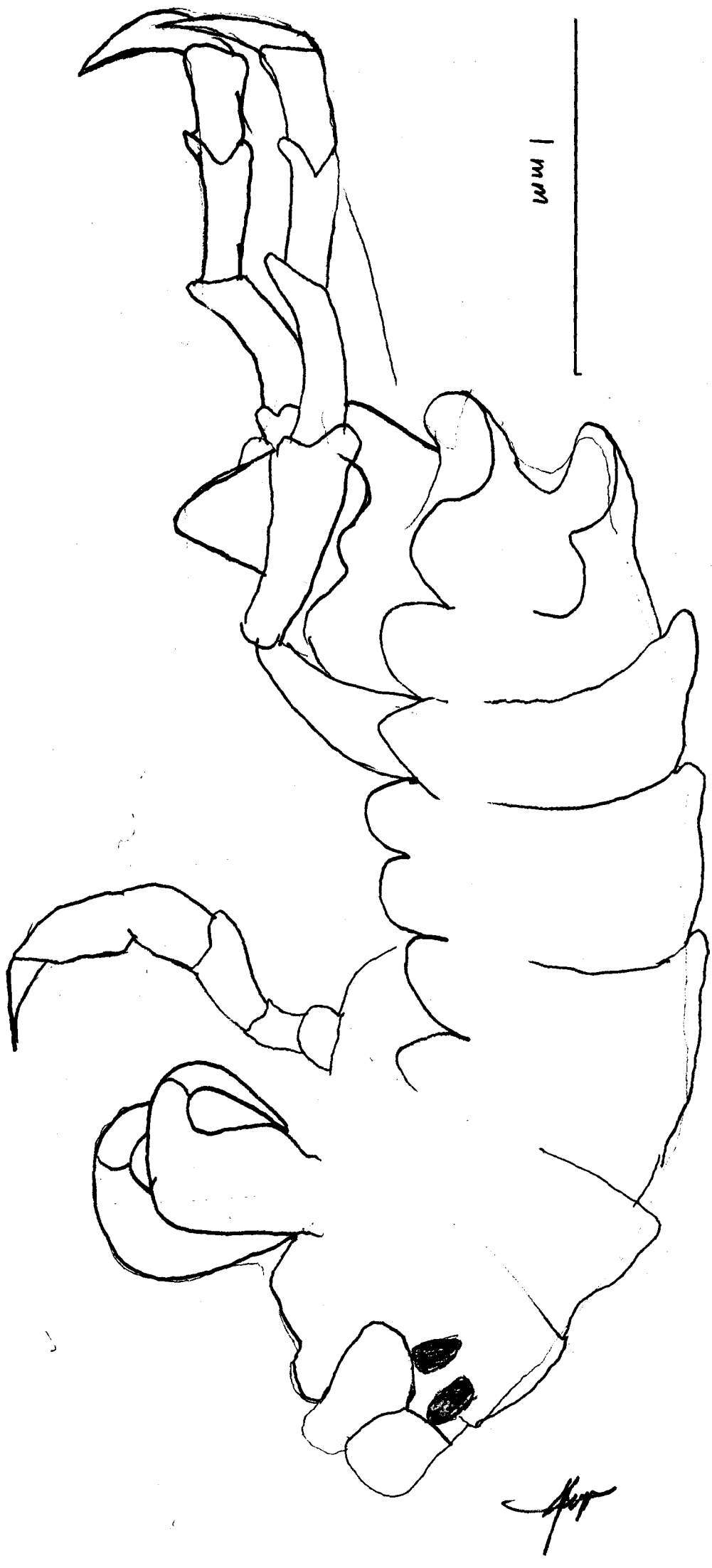


Figure 10:
Podoceridae

1 mm

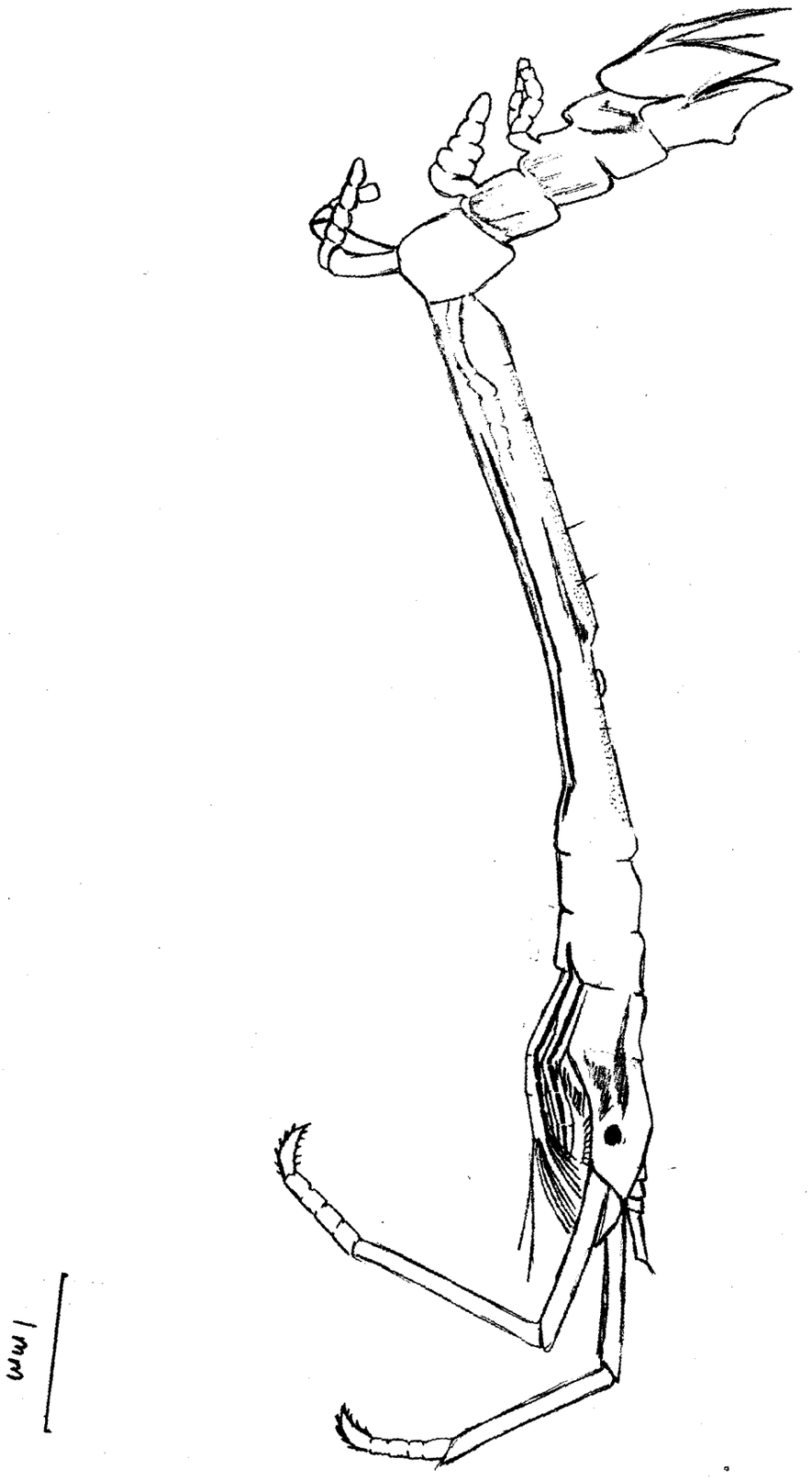
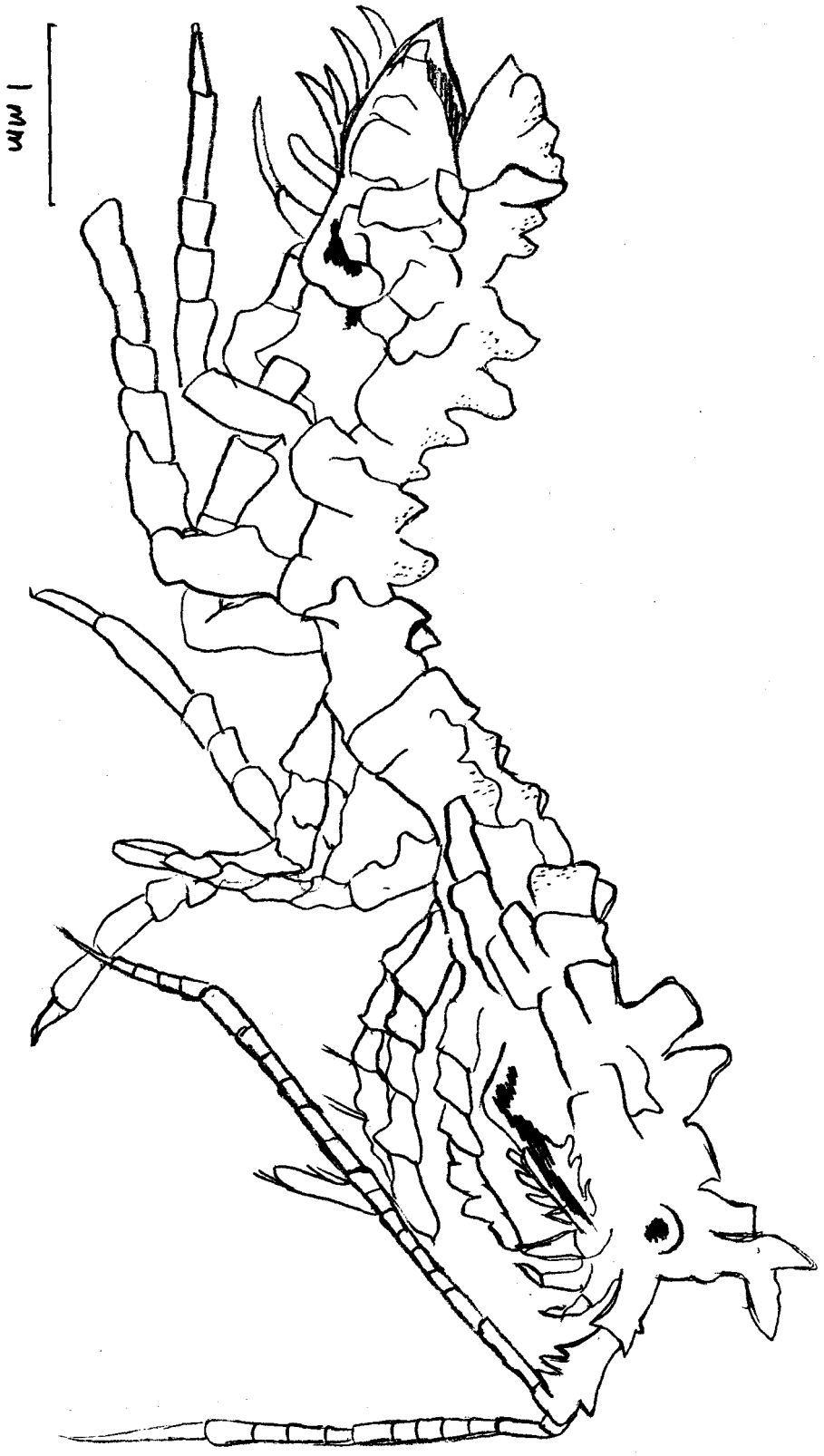
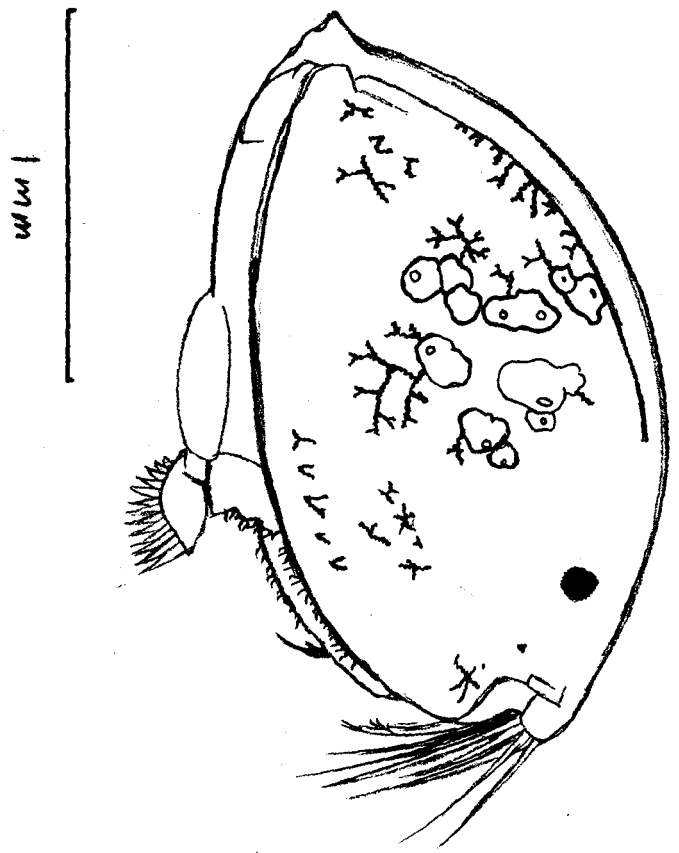


Figure 20. Arcturidae

Figure 21. Arcturidae




1 mm

mm